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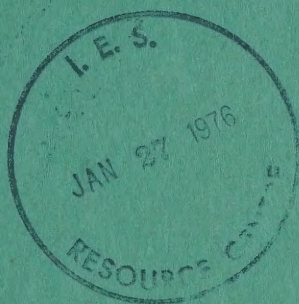
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Mine and Mill Wastewater Treatment



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Report EPS 3-WP-75-5

Water Pollution Control Directorate
December, 1975

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MINE AND MILL WASTEWATER TREATMENT

by

WPCD Staff
in cooperation with the
Treatment Working Group
of the
Mining Regulations Task Force

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ENVIRONMENT CANADA

Report No. EPS 3-WP-75-5
December, 1975

ABSTRACT

This report reviews the origin of water borne wastes (mine drainage, mill process effluent and surface runoff) encountered in metal mining and milling operations, and the associated chemical processes resulting in contaminated effluents. It deals with the types of contaminants normally encountered in wastewaters discharged from the industry and the technology currently employed for the abatement and control of water pollution. The design and operation of the most widely used treatment facility, the tailings pond, is covered in some detail. Treatment of acid mine water in the newer mechanical treatment systems is also reviewed, particularly with reference to the 5 lgpm pilot plant project recently completed at the Brunswick Mining and Smelting Company.

One section deals with the economic impact of effluent treatment on the mining industry. The costs of constructing and operating sound environmental control systems are discussed. The data and method suggested for deriving such costs should prove helpful to professionals involved in the estimation and control of environmental protection costs. Several case studies covering treatment costs are included.

RÉSUMÉ

Le présent rapport traite des origines des effluents produits par l'extraction et la concentration des minéraux métalliques (drainage des mines, effluents industriels, ruissellement), et des processus chimiques expliquant la présence de polluants dans ces eaux. Il traite aussi des types de polluants normalement présent dans les effluents déversés par les industries et des méthodes techniques d'épuration utilisés couramment pour la dépollution et le contrôle de la pollution de l'eau. Il explique quelque peu le principe et le fonctionnement du mode de traitement le plus utilisé de tous, le bassin à résidus. Il s'arrête aussi au traitement des eaux de mine acides par les dispositifs mécaniques les plus nouveaux, en examinant tout particulièrement l'unité pilote d'un rendement de 5 gallons impériaux à la minute, récemment installé à la Brunswick Mining and Smelting Company.

Une partie du rapport est consacrée aux implications économiques du traitement des effluents sur l'industrie minière, notamment aux coûts de construction et d'exploitation. Les données et les méthodes proposées pour l'estimation de ces coûts devraient servir aux responsables de l'évaluation et de la vérification des coûts relatifs à la protection de l'environnement. Quelques cas types portant sur les coûts du traitement des effluents complètent le rapport.

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1. INTRODUCTION

The national Mining Effluent Task Force for the metal mining industry was charged with the responsibility of making recommendations to assist the federal government in its development of regulations and guidelines to protect the environment from mining and milling solid and liquid effluents. To support the activities of the Task Force a number of Working Groups, consisting of technical experts in specialized areas, were formed. One of these was the Mine Wastewater Treatment Working Group. The primary objective of this group was to define "best practicable technology" and the effluent standards which could be achieved through the application of this technology. This report constitutes the data collected and analyzed by this working group for submission to the Task Force. It deals with the sources of water pollution within the mining industry, effluent treatment technology and treatment costs.

2. SUMMARY AND RECOMMENDED EFFLUENT LEVELS

The national Task Force has adopted certain overall criteria as the basis for the development of guidelines and regulations. The first of these is that control should be exercised over the effluent before discharge from an operation and, therefore, the standards developed should be effluent standards.

The second basic principle is that controls should be based on "best practicable technology" to prevent pollution of receiving waters.

It is the purpose of this document to define what the best attainable control levels are, based on the application of "best practicable technology".

The sequence of action taken to achieve these objectives is to:

- 1) Define the basis of the problem with respect to its origins and its detailed characteristics.
- 2) Describe the various unit treatment processes available and assess their application to the problems at hand.
- 3) Describe specific practices currently in use and ascertain what can be considered as "best practicable technology", and the levels which can be attained through the application of this technology for each contaminant controlled.
- 4) Determine the approximate costs of the treatment required to attain these control levels, to confirm "best practicable technology" and determine, in general terms, the impact of these increased costs on the mining industry.

Following are the proposed national control requirements for base metal, uranium and iron ore mines, including mills. These controls will take the form of guidelines for existing operations and regulations for new ones. These control levels constitute a working summary of the findings in this report, as well as those from other Working Groups of the Mining Effluent Task Force.

<u>Parameter</u>	<u>Maximum Acceptable Arithmetic Monthly Mean*</u>
Lead	0.2 mg/l
Copper	0.3 mg/l
Arsenic	0.5 mg/l
Nickel	0.5 mg/l
Zinc	0.5 mg/l
Suspended Solids	25 mg/l
Radium 226	10.0 pCi/l
pH	6.0 or greater

* All concentrations are "total" values except for Ra 226 which is a "dissolved" level after filtering the sample through a 3 μ filter.

3. THE ORIGIN OF WATER POLLUTION PROBLEMS

The sources of water borne contaminants from mining operations can be classified into two groups: acids generated from the exposure of iron sulphide minerals to the atmosphere; and, contaminants which result from the mining and processing of the ore to recover the values and discard the gangue. *6 quanta*

3.1 Acid Generation

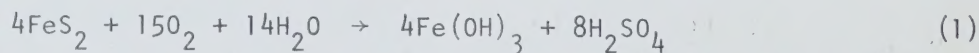
3.1.1 The basic mechanisms

Acid mine waters are produced from the oxidation of metallic sulphide minerals, particularly those containing iron. In theory this oxidation can occur either chemically or through biological means. There are differences of opinion as to the relative significance of the two oxidation mechanisms. In practice, the bacterium *Thiobacillus ferrooxidans* is always present in acid mine waters and there is considerable evidence which suggests that the biological mechanism is significant, if not predominant.

Thiobacillus ferrooxidans is a unique bacterium which obtains its energy for growth from the oxidation of reduced sulphur compounds (i.e. sulphides) and ferrous iron. It is the only organism known that has the ability to oxidize sulphide minerals. In order to perform this oxidation the bacterium requires water, oxygen, carbon dioxide, ammonia, phosphorus and trace amounts of nutrients such as calcium, magnesium, etc. These are usually present in sufficient quantities in natural waters.

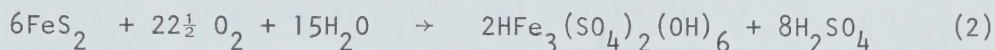
Since the detailed pathway describing the function of *Thiobacillus ferrooxidans* is complex and subject to disagreement, only the net reactions are considered below.

The acid generation process is illustrated by considering the case of pyrite.



This reaction represents the complete hydrolysis of all the ferric iron and the production of two moles of sulphuric acid per mole of pyrite.

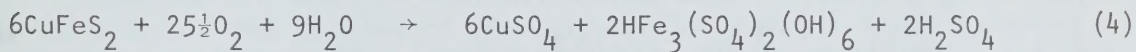
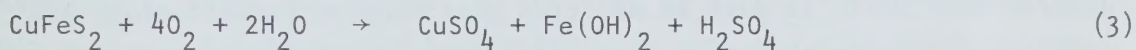
In practice, it is found that not all the iron precipitates as iron hydroxide, but rather as a basic iron sulphate jarosite type material. This may be represented by the formula $AFe_3(SO_4)_2(OH)_6$, where A can be H^+ , Na^+ , K^+ etc. The formation of this material may be represented by Equation (2).



On this basis 1.33 moles of acid are produced per mole of pyrite.

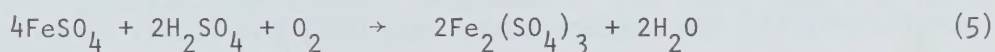
In practice, neither of these equations applies completely and the actual amount of acid produced lies somewhere between 1.33 and 2 moles per mole of pyrite. Further imprecisions arise because, under certain imperfectly understood conditions, significant concentrations of intermediate reaction products of the thiosalt type have been measured.

Similar reactions have been proposed to describe the oxidation of other iron sulphide minerals, e.g. chalcopyrite.

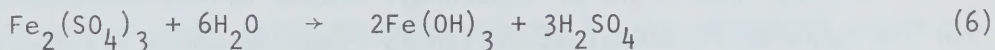


Equations (3) and (4) indicate that acid formation between 1.0 and 0.33 moles per mole of sulphide could be expected.

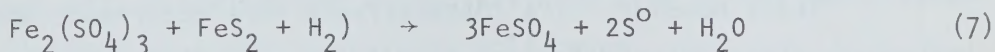
A further set of equations describe the oxidation of ferrous iron.



This reaction occurs in the presence of air.



The ferric ion itself can act as an oxidizing agent on pyrite and thus generate more acid.



3.1.2 Initiation period

Although ideal physical and chemical conditions may exist for oxidation, the reactions do not proceed at their maximum rates immediately. It appears that a gestation or lag period is necessary before significant

oxidation is initiated. This has been attributed to the need to establish a viable population of the bacterium *Thiooxidans* to act as a "trigger" for the reaction. It is also a function of the mineralogy. Pyrrhotite is particularly reactive and will frequently initiate the oxidation of the less reactive pyrite.

The extent of the lag period is variable. Field data suggest times varying from weeks to years may be necessary under different conditions. Factors appearing to influence the lag time include temperature, available surface area, the neutralizing capacity of gangue, and the ferric ion content of the feed water to the tailings.

3.1.3 Factors influencing acid generation

Theoretically, whenever a sulphide mineral is exposed to the atmosphere the potential for acid generation exists. There are numerous interrelated factors which influence the nature and extent of the acid generation. While it must be admitted that many of these factors are imperfectly understood, some generalisations can usefully be made.

3.1.3.1 Inhibition of the oxidation process. The chemical requirements for unrestricted oxidation of iron sulphide minerals have been referred to above and include an adequate supply of water, oxygen, carbon dioxide, ammonia, phosphorus and various micronutrients. The restriction or elimination of the supply of one or more of these will influence acid generation. In the extreme case, this will completely inhibit the process by providing a hostile environment for the bacteria.

The restricted or inadequate supply of micronutrients may account for the apparently anomalous instances reported where all the basic chemical conditions would indicate acid generation yet none is found in practice. The availability of phosphorus may also fall into this category.

This, however, is a comparatively rare occurrence and, as indicated, these materials are generally in sufficient quantities in natural waters. It is not considered feasible to inhibit the oxidation process by the conscious control of these materials. On the other hand, considerable effort has been directed towards the conscious creation of a hostile environment for the bacteria with the objective of eliminating or substantially reducing the problem at source. These have included:

(a) Restricting the supply of oxygen

Oxygen is the primary source of oxidizing potential in the acid generating system although it may operate through indirect routes as in Equations (5), (7), (8). Oxygen enters a mine or a tailings area both as a gas phase and as a dissolved component in the water. The latter can supply only a limited amount of oxygen, since 8-9 ppm of oxygen in water is the saturation level. This corresponds to a maximum of 15 ppm of acid (H_2SO_4) according to Equation (1). Thus, since much higher acid concentrations are experienced than would result from the oxygen content in water,

The oxygen in an underground mine cannot be restricted on a practical basis at this time. Experiments in the United States using a non-oxidizing mining atmosphere and providing the work force with a separate life support system have not demonstrated this technology. Thus acid mine water will be generated during the active life of the mine. After abandonment, seals can be used to restrict both oxygen and water entry. This approach has been used successfully in some old coal mines in the United States. The sealing of the surfaces of open pit mines is still not fully developed and those pits which generate acid are a major unsolved problem in the future development of control technology.

Restricting oxygen access to stored tailings is a much more viable technique at this time. The methods used primarily seek to control the long term acid seepage from a waste embankment. The techniques available are as follows:

i) Revegetation:

An active soil layer consumes oxygen in its uppermost areas and may gradually become anaerobic if enough soil is available. The addition of sewage sludge or other decaying organic matter not only helps consume oxygen but also aids in the rapid establishment of a self-sustaining soil cover.

ii) Flooding:

Water is an excellent diffusion barrier for oxygen. The depth of water needed in any given case is a function of the size of the pond and the wave or mixing action taking place. This technique does, however, require an impervious embankment. It

is also not applicable as a long term control in areas where evaporation exceeds precipitation.

iii) Sealants:

Chemical sealants have been used with some success. Typically, an aqueous solution of the sealant is sprayed onto the surface of the tailings which, when cured, forms a semi-flexible impermeable mat. The tailings material is incorporated into the mat. The sealant will often be mixed with fibrous material to improve its binding properties. Unfortunately, the sealant has a finite life and must be renewed, generally on a two to three year cycle. Access must also be limited if the seal is to retain its integrity. Further, differential settling of the tailings can fracture the seal, leading to the penetration of oxygen. Some surface regrading is usually needed to prevent unnecessary ponding from taking place.

Certain mixtures of iron sulphide minerals naturally possess the ability to form a hard, largely impervious crust of oxidized material which has the benefit of self renewal. It does not, however, appear to be completely impermeable to oxygen.

Impermeable seals of polyethylene, butyl rubber, etc. overlain with approximately 6 inches of tailings have been used. The viability and long term benefits of this approach have yet to be established and this approach is also inherently expensive.

(b) Restricting the supply of water

Techniques for restricting the contact of water with iron sulphide minerals are applicable only to surface aspects of a mining property. The natural, unavoidable seepage of groundwater into a mine and the use of water for drilling, dust suppression, etc. precludes any practicable means of eliminating the contact between water and sulphide minerals.

On the surface it is imperative that uncontaminated surface waters be diverted from areas where they could pick up the products of oxidation. The subject of segregation and diversion of surface waters is considered elsewhere in this report.

The contact of water with sulphide minerals can, however, be reduced in tailings impoundments, waste rock dumps etc. The application of chemical and physical barriers for reducing contact with oxygen, as already discussed, would generally be applicable to reducing contact with water.

The predominant technique employed in Canada is to provide a vegetative cover. It is theorized that, with a good cover and appropriate contouring, precipitation will report largely as agricultural type runoff or be lost through transpiration to the atmosphere. Vegetation does not provide an impermeable barrier and some water will pass through. Substantial problems also exist with respect to establishing a self-sustaining vegetative cover on many acid generating materials. The addition of neutralizing agents, plant nutrients and complexing agents to inhibit the action of toxic substances is frequently necessary.

It has been suggested that, during the last phase in the active life of a property, only inert tailings should be discharged to the tailings impoundment. If these were appropriately distributed they could provide a satisfactory base for the establishment of vegetation. This could have a fairly wide application but would be restricted by the proportion of such minerals in the ore and be contingent on satisfactory plans to deal with the discarded sulphide fraction. In all cases, reclamation should commence as soon as active operations cease in order to take advantage of any initiation period prior to the onset of acid generation.

(c) Isolating sulphide components

Acid generation can only occur if all necessary elements occur at the same time and place. In some cases it is practical to isolate reactive components from air and water. It is undesirable to construct the embankment itself from reactive materials since this optimizes the exposure to air and to water percolating through the system. Separation of sulphides in the tails is not generally possible but may offer a good solution in specific cases. The disposal of the small volumes of concentrated sulphides must be evaluated on a case by case basis. The most reactive tailings components should be buried under less reactive and preferably less permeable components to minimize oxidation.

(d) Reducing the ferric iron

The action of the ferric iron in back triggering the oxidation of pyrite is given in Equation (7). If this process is to be prevented, then contact between the two species must be eliminated. This is clearly not possible when oxidation is occurring as they will both be present in the system. It is, however, possible to direct ferric iron bearing waste-waters away from iron sulphide minerals. As ferric iron can be present in mine water, surface drainage and seepage water, such waters should not be directed to a tailings impoundment unless they have been treated to render the ferric iron innocuous. This is generally accomplished by neutralization to form a ferric hydroxide precipitate before the waste-water is discharged to the tailings pond. Alternatively, it could occur in the upper strata of the stilling basin in the tailings impoundment. This latter course is less precise and lacks control.

The preferable route is to effect the conversion in a treatment system totally divorced from the tailings impoundment.

Details of the available treatment processes are considered later in this report.

(e) Controlling pH.

The influence of pH on the oxidation process is complex and imperfectly understood. The following hypothesis is suggested.

- i) Although the activity of the bacterium *Thiobacillus ferrooxidans* is reduced at high pH values the bacterium still exists, although possibly in a dormant state. In the absence of sufficient buffering capacity at elevated pH values the oxidation process will ultimately take place and result in acidic conditions. Thus, while it is acknowledged that high pH values probably influence the kinetics of the reaction, they do not eliminate the oxidation process.
- ii) At pH values in excess of 4.0 to 4.5, ferric iron is in the form of the hydroxide and thus not available to back trigger the oxidation reaction.

iii) Alkalinity is, almost without exception, applied in the form of lime or limestone. In reacting with any acid produced in the oxidation process, insoluble calcium sulphate and/or jarosite type materials are produced. These coat the reactive surface of the sulphides and render them impervious to oxygen and water, thus inhibiting the reaction at source.

While not denying the possibility of other contributory mechanisms it is contended that points (i) and (ii) are the dominant factors in the inhibition of the oxidation reaction.

It would be possible to take advantage of this inhibitory mechanism in short term through the dissemination of limestone in the tailings and waste rock. It may be necessary to make the application only in the upper strata and rely on other factors (e.g. limited diffusion) to restrict the passage of oxygen to potentially reactive particles contained at depth. The long term inhibition potential is imperfectly understood and subject to revision.

(f) Use of bactericides

Various synthesized organic bactericidal agents have been tested. At this time they do not provide a practicable solution to the inhibition of the oxidation process.

(g) Limiting the area of the reactive surfaces

The surface area participating in the oxidation process has a substantial effect on the kinetics and yield (per unit weight) of the reaction. Thus, for a given mass of iron sulphide minerals, the finer the particles the more rapidly oxidation would occur. This does not discount the results of oxidation of coarse materials. While the apparent surface area of such materials may be comparatively low, they frequently exhibit significant internal areas available for reaction through stress fractures and crystallographic planes. An overall fine size distribution may also reduce percolation rates. The net effect of these competing elements must be evaluated on a specific basis.

(h) Controlling temperature

The kinetics of oxidation using the bacterium *Thiobacillus ferrooxidans* are strongly temperature dependent. The optimum temperature

for the reaction is 35°C. At temperatures of less than 5°C the rates of reaction are extremely low. This may account, in part, for the lack of acid conditions reported north of latitude 60° although it has no apparent effect on the metal mining areas of Northern Manitoba and Ontario.

Ice formations may effectively prevent seepage from waste rock dumps and, more rarely, tailings areas. Recently it has been suggested that ice plugs formed via venting cold winter air through abandoned underground workings may provide low maintenance seals to reduce the discharge of acid waters from such properties.

3.2 Contaminants Arising from the Processing of Ores

3.2.1 Milling reagents

The processing of ores to recover the valuable minerals involves a wide range of unit operations. These exploit differences in physical and chemical properties of the different minerals to effect separations. These unit operations are described in the report from the Water and Waste Minimization Working Group (1). Many of these operations involve the addition of reagents which modify a property of a particular mineral or class of minerals. A comprehensive review of the toxicity of a wide variety of these milling reagents has been prepared by Hawley (2). For detailed information the reader is referred to that report. For the sake of completeness some of the basic findings are repeated here:

- 1) Mine-mill reagents vary widely in their toxic effect. Some are persistent and hence will escape from a tailings area while others are unstable and will break down in a tailings area. Obviously, when reagents are being chosen for use in a mine-mill circuit, the least toxic compounds available should be chosen. Included in the definition of 'least' toxic compounds are those toxic compounds that break down rapidly (into innocuous substances) in a waste stream, or those toxic substances which can be removed easily from a waste stream using accepted waste treatment technology.

- 2) The use of mine-mill reagents that are persistent and toxic, or reagents that do not break down easily in a natural environment, should be avoided if possible.
- 3) The use of mine-mill reagents having known or suspected nutrient properties should be avoided if possible.
- 4) When choosing a reagent for use in a mine-mill circuit or process, the total effect of that particular reagent on the receiving watercourse should always be considered. Reagent characteristics to investigate, for instance, would generally include chemical oxygen demand, biochemical oxygen demand, biodegradability, effects on local aquatic life (plant and animal), and effects on the total dissolved solids concentration and hardness of the receiving stream.
- 5) The use of mine-mill reagents that consist of, or contain, water-soluble salts of metals that are known to exhibit undesirable environmental effects should be avoided if possible.
- 6) Only occasionally will any commercial reagent exist in the effluent from a typical tailings area in concentrations greater than 3 parts per million. The most common concentration of a commercial product in a typical tailings area effluent appears to be 2 parts per million or less.

Residual concentrations of mine-mill reagents have been identified in the effluents from mining operations that work sulphide ores. Xanthate residuals of 0.2 to 1.2 mg/litre and dithiophosphate residuals in the range of 0.3 to 2.7 mg/litre have been noted. In one case, isopropyl ethylthionocarbamate was detected in an effluent at a concentration of 1.8 mg/litre.

- 7) In general, frothers are quite volatile and undesirable in appearance. Their build-up in the tailings water, particularly for use as recycle water to the mill, can probably be reduced or eliminated by:
 - (a) increasing the retention time of the frother-containing wastes before recycle to the mill;
 - (b) applying a degree of mechanical aeration to the frother-containing waste; and,

- (c) selecting another frother with superior breakdown properties for use in the mill.

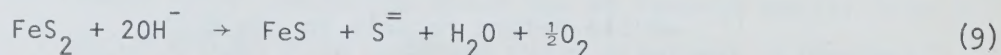
If the breakdown properties of a particular frother are known, then the actual waste retention time in the tailings area serving the operation can probably be increased to effect substantially a complete breakdown (or loss) of the frother, or decreased to prevent frother loss or breakdown (if this is desired for recycle).

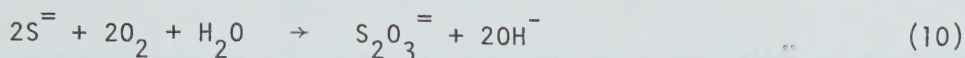
The xanthates are among the most toxic reagents that find common use in a sulphide mill circuit. They are, however, quite unstable in aqueous solution and break down into relatively innocuous substances. At temperatures above 30°C and at low concentrations (such as found in a tailings decant) and exposure to acidic conditions, decomposition of xanthates is extremely rapid. Therefore, it appears that residual xanthate concentrations can be reduced by increasing waste retention times and by decreasing solution pH values.

3.2.2 Thiosalts

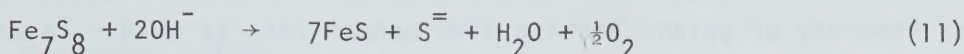
The presence of thiosalts in mining effluents can be attributed to two basic sources. First they can originate from the milling of sulphide ores, and second, from the use of reducing sulphur compounds in flotation circuits.

3.2.2.1 Polythionates from grinding operations. During grinding and flotation operations a small amount of sulphur from the minerals dissolves in the water. The process seems to be independent of water temperature but is known to accelerate in alkaline solutions normally used in some milling operations. In one particular operation, the water leaving a typical mill has a chemical oxygen demand (COD) of approximately 50 milligrams per litre. This is not particularly high but the products of oxidation with the passage of time generated sulphuric acid which caused the pH of the soft receiving water to decrease to approximately three units. The sulphide conversion to thiosulphate in the mill circuit is described in the following equations:





Similarly, when pyrrhotite predominates in the ore the following reaction may occur:

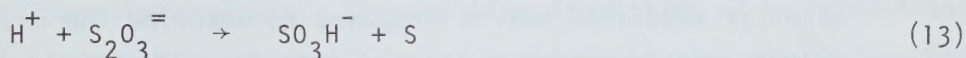


this is also followed by reaction (10).

The following is an attempt to explain how a mixture of sulphur-oxygen compounds can be produced in the grinding and flotation circuits although the exact reactions are not known and other sources proposed slightly different routes. Presumably, some of the sulphur is sheared off the ore in elemental form which, in ambient conditons, exists as a staggered eight member molecular ring.



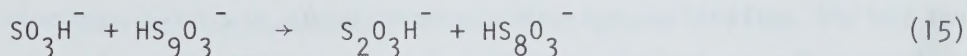
The free sulphur reacts with the sulphite ion which is produced in solution by dissociation in accordance with reaction:



and the combination of free sulphur and sulphite produce:



This product is a linear molecule of sulphur-sulphur multiple bonds which is highly assymetrical and open to further S_N^2 attack by the SO_3H^- ion acting as a nucleophile.



This reaction repeats itself and the resulting solution will contain a series of linear sulphur molecules, $S_3O_6^{=2}$, $S_4O_6^{=2}$, $S_5O_6^{=2}$ etc., which are termed generally as polythionates.

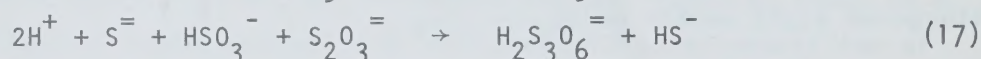
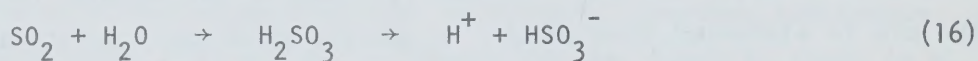
Polythionates could also be created through the incomplete oxidation of iron sulphide minerals.

3.2.2.2 Effects of reducing sulphur compounds. In certain ore milling applications, reducing sulphur compounds must be used to enable selective flotation of certain minerals. These compounds are generally in the form of alkaline sulphides, hydrosulphides, sulphites and sulphur dioxide.

Marginal contributions may be made through the breakdown of xanthates and dithiophosphates.

As an example, sulphur dioxide is useful in depressing marmatite for recovery of galena bearing zinc concentrate at pH 4. In another application, sulphur dioxide is used for separation of copper-lead minerals from zinc bearing ore at pH 4.

The use of sulphur dioxide results in increasing polythionate concentrations. This can be explained through Equations (11), (12), (13), (16) and (17).



It is expected that mills which use sulphur dioxide could have polythionate concentrations of more than 1,000 milligrams per litre in the wastewater.

Similar reactions may be proposed to describe the oxidation route of other reducing sulphur compounds.

3.2.2.3 Effects of water recycling on polythionate concentrations. The rate of dissolution of sulphide ions from ore particles, governed by reaction (9), is proportional to the alkalinity of the milling water, the length of time the ore particles are exposed to high pH in the mills and flotation circuits, the water temperature and the degree of agitation. When the pH, agitation and water temperatures are held approximately constant, the longer the period of exposure of water to the minerals, the more polythionates can be expected in the water. Each time the water is recycled, it is re-exposed to the ore, prolonging by several times the period of exposure in comparison to water that is used only once. Therefore, it is very likely that the concentrations of polythionates in both types of mills which recycle water, those that use reducing sulphur compounds as well as those that do not, will increase. Although the effect of polythionates on the selectivity of the flotation process cannot be forecasted, recycling may have some beneficial effects, such as savings in reagents.

It is unlikely that the total polythionate loading from mills practicing total recycling will be any higher than from mills in which

water is used only once. The volumes discharged to the environment will be smaller giving some very obvious benefits, such as lower investment in equipment to treat the water and possibly lower reagent consumption.

3.2.3 Suspended solids

In order to recover the valuable constituents, ore must be finely ground. The liberated mineral particles are recovered in the milling process and the waste gangue is rejected as mill tailings. The controlled deposition of the tailings, generally in an engineered impoundment, is a traditional practice for Canadian mining companies. The water used to transport the tailings to the impoundment is freed after settlement of the particulates and discharged through a weir or some other form of decant structure for recycle to process or discharge to the environment.

In recent years, finer liberation sizes and increased backfilling practices have resulted in substantially finer tailings being discharged to the tailings impoundment. The settling properties of colloidal and near colloidal sized particles pose special problems if they are to be effectively retained in the tailings impoundment and not report in the decant. The use of flocculants to assist in settling the solids in the tailings pond may be necessary if the pond size is not sufficient.

3.2.4 Radioactive substances

In the uranium mining industry an additional problem lies in the presence of radioactive compounds in the mining and milling effluents. Of primary concern to date has been the presence of Radium 226, but there are other radioactive compounds such as Thorium 230 and 232 and Lead 210 which may be present. Radium 226 is naturally present in very small concentrations in all uranium bearing ores. Most of the radium remains in the undissolved states throughout the mining and milling operation, but a small amount of the radium in the ore (about 1 percent) does dissolve and find its way into the tailings effluent.

When considering wastewater treatment at a mine/mill operation three main types of effluent must be included: mine water discharge, mill process effluents, and surface drainage. Mine water, in the case of Elliot Lake mines, contains sufficient uranium values to make it advantageous to

pump it to the mill for uranium recovery. The mine water then reports in the mill process effluents.

The data contained in Table 1 illustrate the range of contaminants for various water flows found in a uranium mining operation. These data illustrate the need to treat process streams and contaminated surface and seepage waters. The methods used are discussed in Section 4. There are two sets of numbers listed for some parameters (filtered and unfiltered). This reflects the historical practice within the industry of using filtered analyses only. Thus, there are a limited amount of data using unfiltered analyses, which is the method of choice within the context of environmental protection. The industry recognizes this deficiency and has proposed a regular monitoring program for at least one year to measure the levels of Radium 226, Thorium 230 and 232, and Lead 210 using both methods. This program will be supported by various government laboratories capable of conducting these assays.

It is premature to draw final conclusions as to which radioactive products are of real concern but it appears possible that all products other than Radium 226 can be safely eliminated for the purposes of environmental monitoring and control. It is recognized, however, that all discharged radioactive materials should be reduced to the lowest levels possible using the best technology available.

TABLE 1. CHEMICAL AND RADIOACTIVE PARAMETERS FOR URANIUM MINES.

(F = Sample Filtered on 0.45 or 3.0 μ filter; UF = unfiltered)

Water Type	pH	Ra 226 pCi/l	Total Thorium pCi/l	Uranium mg/l	Pb 210 pCi/l
Mine Water (F)	2 - 4	7-170	5-17,000	0.1	---
Surface Runoff (F)	2 - 4	15	7,000	3	---
Seepage (F)	2 - 4	1-12	3-10,000	1-8	---
Ion exchange Barren (F)		2-3,000	70,000 to		1-4,300
(UF)		2-4,000	360,000		1-4,800
Feed to BaCl ₂ Treatment (F ²)	6 - 9	200-800	20-40	1	3-7
(UF)	6 - 9	270-900	30-100	--	10-18
Decant after BaCl ₂ Treatment (F)	6 - 9	2-8	10-40	0.15	2-8
(UF)	6 - 9	50-200	10-100	--	3-10

4. WASTEWATER TREATMENT PROCESSES

The specific wastewater problems and treatment requirements of a particular mine are subject to the influence of local circumstances. These include the mineralogy of the ore body and surrounding area, climate and topography. However, the overall problem faced will be composed of an aggregate of discrete units determined by the contaminating factors described in Section 3.

In this section, descriptions are provided of the basic wastewater treatment processes. Emphasis is placed on demonstrated, practicable solutions applicable to Canada.

4.1 Acid Neutralization and Heavy Metal Removal

Acid is generated through the oxidation of iron sulphide minerals. This acid is then available for leaching metals from the minerals.

The characteristics of a range of typical acid waters is given in Table 2. In their untreated form, these wastes are extremely toxic to fish and other forms of aquatic life and do not comply with many basic requirements for potable water. Any process used to neutralize acids will also remove dissolved heavy metals through the formation of insoluble metal hydroxides. The conditions necessary for hydroxide formation are considered below.

4.1.1 Hydroxide formation

Heavy metal precipitation occurs as a result of the reaction $M^{++} + 2OH^{-} \rightarrow M(OH)_2$, where M is the metal cation. Many metal hydroxides are in fact precipitated as the hydrated metal complex $M(OH)_2 \cdot nH_2O$, even from simple solutions of the metal.

The precipitation of heavy metal hydroxides can be predicted by using standard solubility relationships where the solubility product, K_{sp} , for the above general reaction is given by:

$$K_{sp} = (M^{++}) (OH^{-})^2$$

Since the OH^{-} ion concentration is a function of hydrogen ion concentration, the degree of precipitation of a given metal hydroxide is directly related to pH:

$$(OH^{-}) = 10^{(pH-14)}$$

TABLE 2. TYPICAL ASSAYS OF ACID WATERS IN CANADA

(All Concentrations are mg/l except pH)

Type of Operation	Cu - Pb - Zn (Mine & Surface Drainage)	Cu- Pb -Zn (Mine Water)	Uranium (Seepage)	Cu - Zn (Active Mine)	Base Metal (Abandoned)	Uranium (Abandoned Mine)
pH	4.0	2.0	2.0	3.0	2.6	2.0 - 2.8
Suspended Solids	8.8	690	Nil	-	-	25
Total less solids	79	24,000	-	-	9,200	13,440
Hardness	293	2,960	-	-	1,390	
Ca	-	-	416	-	454	-
Mg	-	-	106	-	178	-
Cu	17	11	3.6	0.0	2.5	2.2
Zn	118	1,090	11.4	0.4	34	9.4
Pb	0.4	58	0.7	0.11	0.5	-
Fe (total)	79	1,830	3,200	11.7	1,300	300
Mn	21	0	5.6	0.4	8.2	3.6
SO ₄	36	16,560	7,440	885	4,050	6,900
COD	-	245	270	-	110	-

Thus, for simple precipitation, if the solubility product constant is known, a plot of metal ion concentration against pH can be drawn from which the optimum level of precipitation and corresponding pH can be determined.

The calculations are more complex if the precipitated compound is amphoteric or if the composition of the waste is such that several compounds precipitate simultaneously.

Amphoteric precipitates, such as many heavy metal hydroxides, exhibit minimum solubility at a distinct pH level and an increase in solubility as pH is shifted in either direction. The pH level of minimum solubility can be evaluated from solubility product and chemical equilibrium relationships of the appropriate ampholytes. Such an exercise has been attempted for copper, zinc, lead and iron and the resulting plots are presented in Figures 1 to 4.

Several factors preclude the possibility of theoretically deriving absolute values for best residual levels of metal ions. Not the least of these is the extremely broad range of solubility products for most of the metallic salts in question.

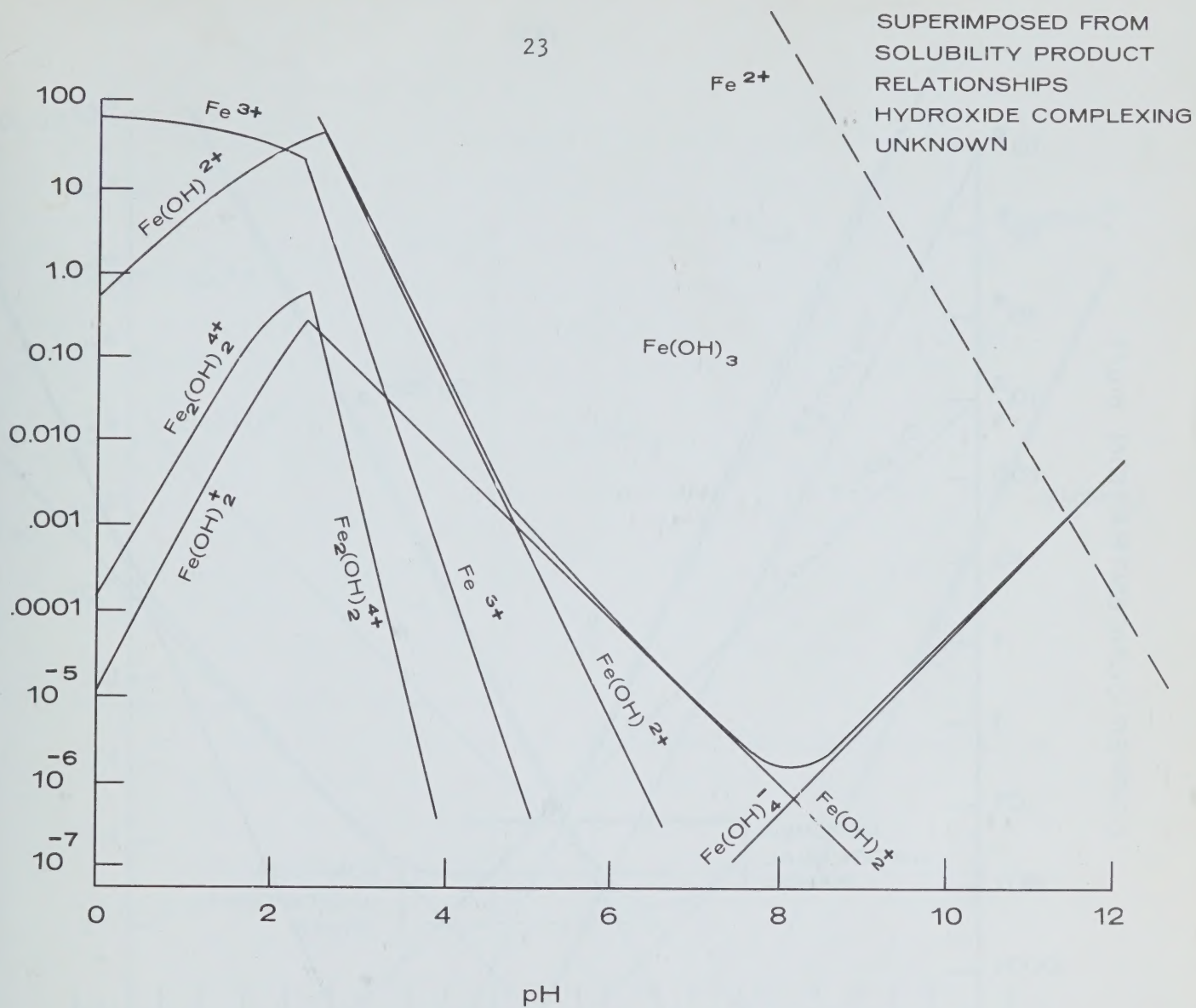
Bearing in mind the above-mentioned limitations, the following ranges of residual metal concentrations and optimum pH values appear to be theoretically feasible if total precipitate removal from the decant can be achieved.

Cu	1 - 8 $\mu\text{g/l}$	pH 9.5
Zn	10 - 60 $\mu\text{g/l}$	pH 10
Pb	< 1 $\mu\text{g/l}$	pH 8
Fe (total	< 1 $\mu\text{g/l}$	pH 8 (if totally oxidized to ferric)

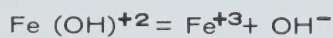
It is evident that, at best, theoretical solubility product relationships can give only a broad indication of the metal concentrations attainable by treatment methods using precipitation.

Figures 1 to 4 indicate that the following pH sequence should theoretically result in an optimum removal of metals from solution:

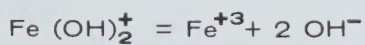
pH 8 - maximum removal of iron as $\text{Fe}(\text{OH})_3$ and
lead as $\text{Pb}(\text{OH})_2$



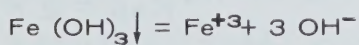
REACTIONS:



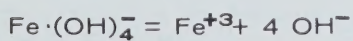
$$K = 1.5 \times 10^{-12}$$



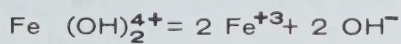
$$K = 5.6 \times 10^{-22}$$



$$K_{\text{sp}} = 10^{-38}$$



$$K = 10^{-33}$$

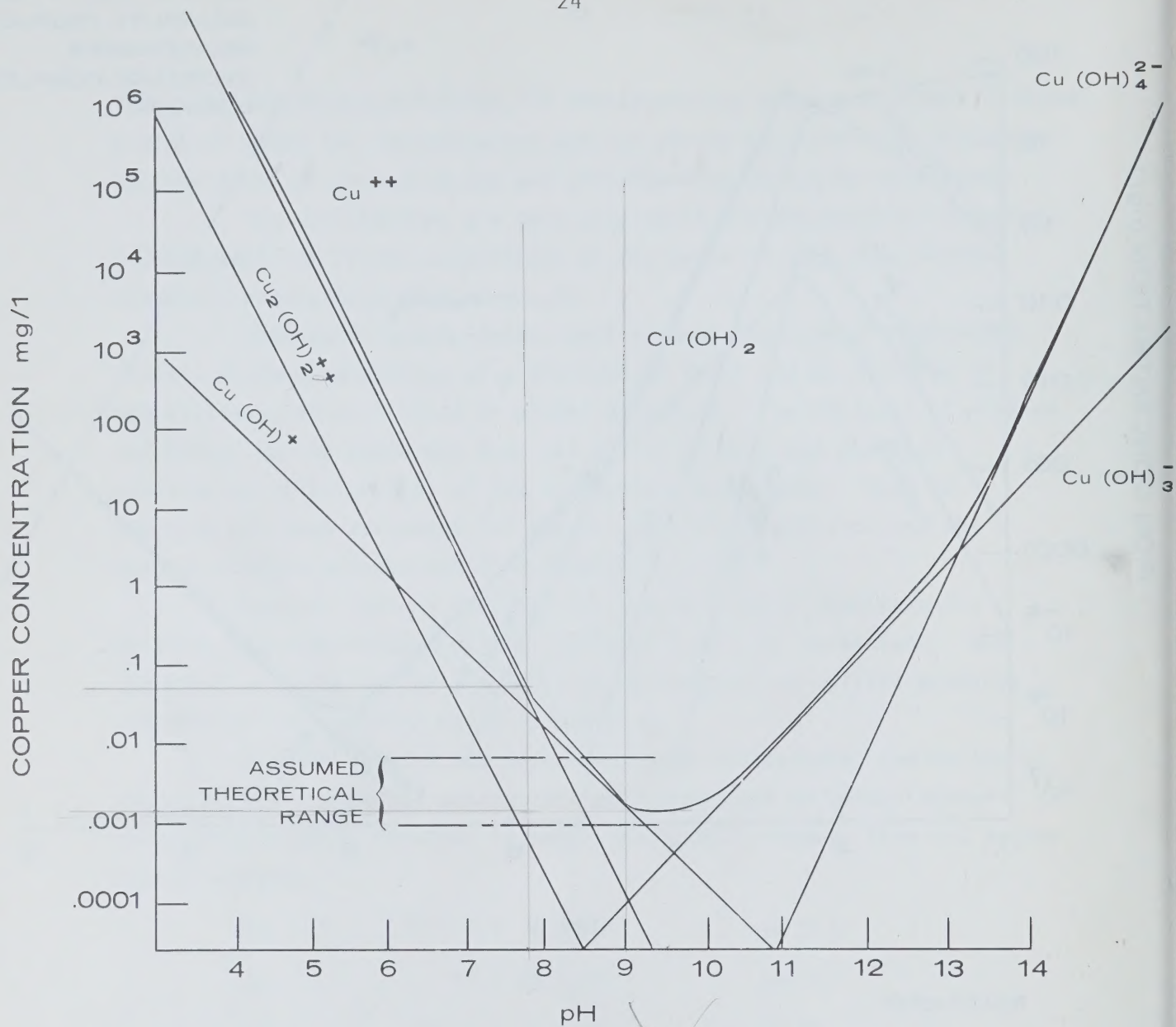


$$K = 7 \times 10^{-26}$$

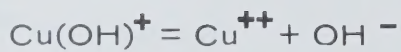


$$K_{\text{sp}} = 3.2 \times 10^{-14}$$

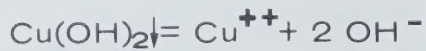
Figure 1. Iron Solubility (3)



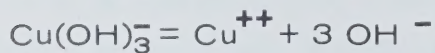
REACTIONS:



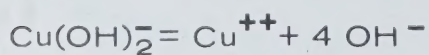
$$K = 10^{-6}$$



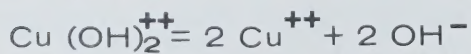
$$K_{sp} = 2 \times 10^{-19}$$



$$K = 6.2 \times 10^{-16}$$

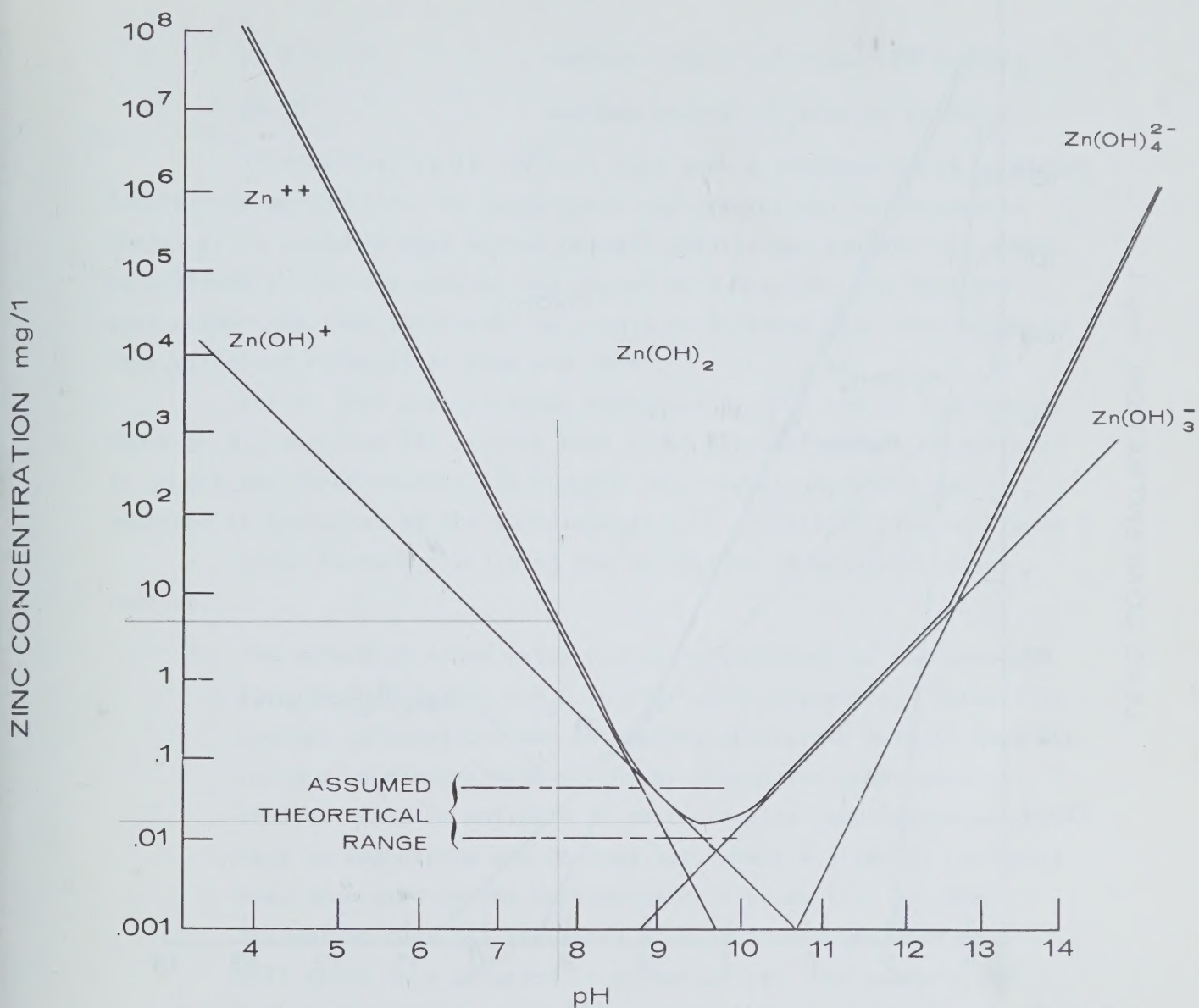


$$K = 7.7 \times 10^{-17}$$

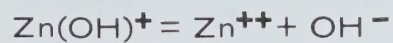


$$K = 10^{-17}$$

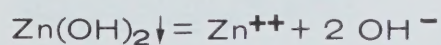
Figure 2. Copper Solubility(3)



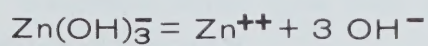
REACTIONS:



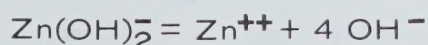
$$K = 9.1 \times 10^{-6}$$



$$K_{sp} = 4.5 \times 10^{-17}$$

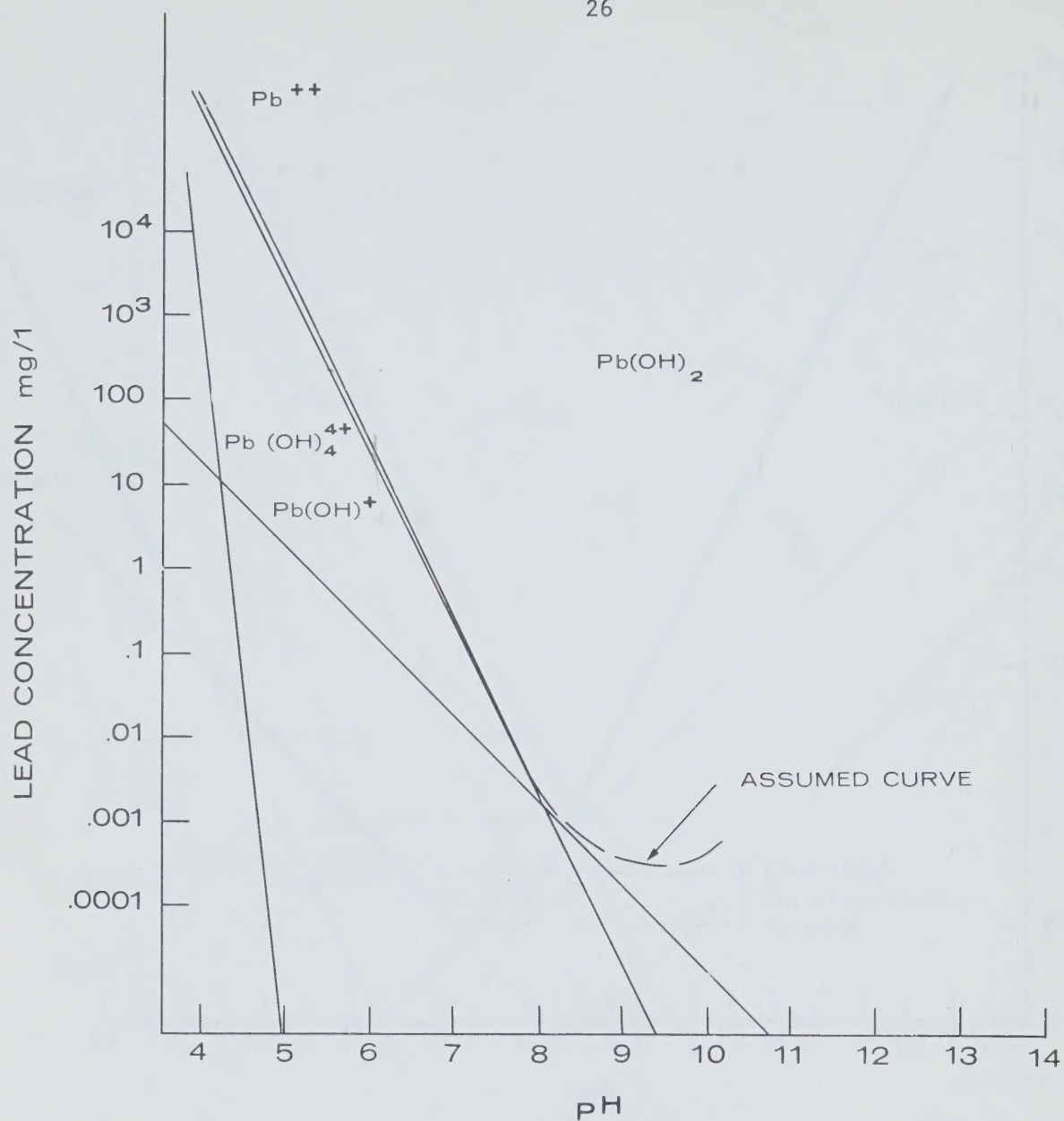


$$K = 1.75 \times 10^{-14}$$



$$K = 8 \times 10^{-16}$$

Figure 3. Zinc Solubility (3)



REACTIONS:

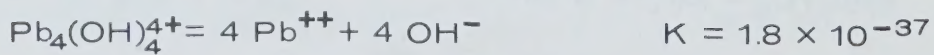
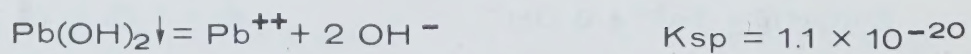
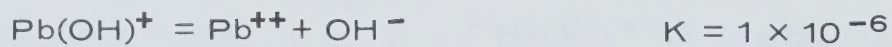


Figure 4. Lead Solubility (3)

- pH 9 - 9.5 - maximum removal of copper as $\text{Cu}(\text{OH})_2$
- pH 10 - maximum removal of zinc as $\text{Zn}(\text{OH})_2$

In practice, it is unlikely that such a sequence would be either feasible or worthwhile. In cases where the theoretical relationships apply, it is probable that better overall results and reliability would be achieved by closely controlling the pH at 9.5 to 10. In this way good copper and zinc precipitation should be achieved with only slightly less efficient removals of iron and lead.

Ferric iron precipitation commences at pH 3 and is complete by about pH 8. Harrison (4) reports that total ferrous removal is achieved at pH 9.5 but other sources (3) suggest that higher pH values may be required as indicated by the superimposed Fe^{++} solubility line on Figure 1.

Other factors precluding the derivation of absolute values include:

- The effect of mixed metal hydroxide complexes of the general form $(\text{M}_1)_a(\text{M}_2)_b(\text{OH})_c \cdot n\text{H}_2\text{O}$ is difficult to predict. Solubility product calculations can be applied accurately only if the ratio of concentrations of the precipitating substances is stable. This is unlikely in most practical applications.
- Data on ampholytes are limited but, where available, such data have been used in the derivation of Figures 1 to 4. The assumption that all indicated ampholytic species of a metal will exist in a solution is conservative. For example, in the case of copper it has been assumed that $\text{Cu}(\text{OH})_2$ precipitates, and $\text{Cu}(\text{OH})^+$, $\text{Cu}(\text{OH})_3^-$, and $\text{Cu}(\text{OH})_4^{=}$ remain in solution.
- Chelation of metal ions by organic compounds in the tailings pond can further influence actual ion concentrations.

4.1.2 Neutralizing reagents

Reagents which have been used for neutralization of acid mine water include:

- limestone,
- lime,
- sodium carbonate, and
- sodium hydroxide.

In the iron ore industry, where flotation is practiced, the circuit is quite often run at a pH of 9 - 11.5. For effective flocculation of slimes in tailings the pH should be neutral. The neutralizing agent most commonly used is sulphuric acid.

The relative costs and advantages of these reagents are summarized in Table 3. This information was developed on materials costs and supplies at a particular location in New Brunswick. Transportation costs, reagent purity, and the availability of materials may provide a different cost analysis in other parts of the country. It is evident that on a cost basis alone, limestone and lime are the most attractive, with limestone being the cheaper when compared in terms of relative basicities. The effect of dolomite content has not been considered but can be a major factor. Several serious disadvantages are associated with the use of limestone, particularly its inability to produce pH's higher than about 7 and its tendency to produce a surficial layer of gypsum (CaSO_4) and gelatinous metal hydroxides, which reduces its reactivity unless mechanically removed by sparging or tumbling. Lime is, therefore, the most common reagent used for pH adjustment in base metal mine effluent treatment and results in the metal being precipitated as the hydroxide.

In certain instances it is advantageous to consider the use of mixtures of limestone and lime. In such cases limestone is applied initially with final pH adjustment made with lime. The factors which have to be assessed include:

- (i) unit reagent costs and daily tonnage requirements;
- (ii) availability of supply;
- (iii) reaction period necessary for neutralization
(limestone can require up to 48 hours contact before the reaction is complete);
- (iv) capital and operating costs of dual materials handling system;
- (v) sludge handling and stability;
- (vi) process control; and,
- (vii) freight costs for reagents.

TABLE 3. ALKALI CHARACTERISTICS

Reagent	Formula	Basicity Factor*	Approximate Cost (¢/lb)	Relative Cost/ Unit Basicity (cents)	Relative Advantages and Disadvantages
Limestone	CaCO_3	1	0.4	0.4	Suitable for pH < 7 only. Lower reactivity than lime. Potential surface coating problems with gypsum. Handling costs may be relatively high - crushing, high bulk storage, etc. High CO_2 production.
Lime	Ca(OH)_2	1.75	1.25	0.71	High reactivity - relatively easy bulk handling, high pH attainable.
Ammonium Hydroxide	NH_4OH	2.85	8	2.80	Nitrogen nutrient enrichment has eutrophication implications in some regions. NH_3 toxic to fish at high pH and low DO.
Sodium Carbonate	Na_2CO_3	0.91	3	3.30	Less basic yet more expensive than limestone or lime. High CO_2 production.
Sodium Hydroxide	NaOH	1.25	5	4.0	Cost prohibitive - very fast reacting, complete pH range effectiveness.

* $\text{CaCO}_3 = 1$ (100% utilization of available alkalinity assumed)

Finally, it must be emphasized that the reactivity and thus the cost/effectiveness of different sources of lime and limestone vary tremendously. Generally, dolomitic limestone and high magnesium lime are less reactive than low magnesium sources.

Reactivity should generally be assessed on the basis of available CaO or CaCO_3 . It is recommended that each application be individually assessed.

It should be noted that metals precipitated by both limestone and sodium carbonate may be in the form of basic metal carbonates; these may be less soluble than the equivalent hydroxides. Sodium carbonate also has the advantage of dispersing slimes that might otherwise float on the surface of the tailings pond and be discharged with the overflow.

The use of ammonia for neutralization is not recommended. Ammonia is an acutely toxic substance even at low concentrations and it is the recommendation of this working group that the use of ammonia for neutralizing acid wastewater be prohibited.

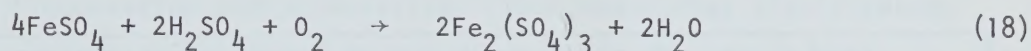
4.1.3 Point of addition

Some basic observations may be made with respect to the point and manner of addition of the neutralizing reagents. Both lime and limestone require a finite reaction time before the neutralizing capacity is fully realized. The required reaction time is very much larger in the case of limestone. This must be taken into account in the original process design, which must establish conditions of intimate contact between the reacting species for as long as is necessary. Clearly this is not achieved if the neutralizing agent is allowed to settle out in a tailings pond or lagoon before it has reacted with the bulk water phase. It may be necessary to provide vessels or use a slurry pipeline or launder as reaction chambers.

4.2 Ferrous/Ferric Oxidation

In order to precipitate ferrous hydroxide a solution pH of 9.5 is required. On the other hand, ferric iron precipitates sufficiently by pH 4.5, although the minimum level as previously indicated is reached at a pH of 8. Thus, there are substantial potential advantages in being

able to oxidize ferrous iron to the ferric form and complete the precipitation at lower pH values. Both chemical and biological oxidation processes have been evaluated. The general equation:



describes the reaction.

4.2.1 Chemical oxidation

Ferrous iron may be oxidized by atmospheric oxygen. The reaction is strongly pH dependent and occurs almost instantaneously (less than 1 minute) at pH values greater than 7.2. At a pH of 6.5 the reaction takes in excess of 1000 minutes. See Table 4.

The importance of the pH has a major cost impact on two inter-related factors: the size of the oxidation reactors; and, the neutralizing agent used for pH adjustment. If the cheaper limestone is used, then a substantial equilibration period is required to approach, if practical, pH values in excess of 7. Thus, large reaction chambers would be required for both neutralization and oxidation. Alternatively, lime could be used to provide rapid neutralization to a pH level sufficient for almost instantaneous oxidation. The attendant implications with respect to sludge production and handling are discussed in Section 4.5.

In the majority of installations treating solely iron bearing acid waters lime or limestone/lime combinations are employed. Different cost evaluations must be applied to waters containing dissolved zinc, nickel, etc. In these cases pH values in excess of 9 will be required to precipitate the metal hydroxides. Consequently, lime handling facilities will be required independent of what material is used to facilitate the iron oxidation process.

4.2.2 Biological oxidation

Biological oxidation of ferrous iron can be effected by the acidophilic bacterium *Thiobacillus Ferrooxidans*. The process consists of controlled bacterial oxidation in a balanced nutrient medium. The process is conducted under acid conditions in the range of pH 1 to 4. The potential advantage of this process is that ferrous iron oxidation is carried out under acid conditions, and thus the converted ferric iron

TABLE 4. DETENTION TIMES (MINUTES) REQUIRED TO OXIDIZE FERROUS IRON
(Temperature 10°C)

pH	100 mg Fe/l	500 mg Fe/l	2500 mg Fe/l
5.0	7.8×10^5	1.1×10^6	1.4×10^6
6.0	7.8×10^3	1.1×10^4	1.4×10^4
7.0	7.8×10^1	1.1×10^2	1.4×10^3
8.0	7.8×10^1	1.1×10^0	1.4×10^0

TABLE 5. FERROUS IRON OXIDATION RATES AT VARIED
INITIAL Fe^{++} CONCENTRATIONS AND TEMPERATURES

Initial Fe^{++} Conc. mg Fe/l	Temperature °C		
	10	20	35
10	0.001*	0.003	0.003
100	0.009	0.013	0.018
500	0.011	0.035	0.053
1000	0.017	0.040	0.080

* oxidation rate g Fe/l/hr

may be precipitated at pH values readily attainable through limestone neutralization. Completely stirred tank reactors (CSTR) can meet process requirements. However, the bacteria do not possess satisfactory auto or induced flocculation characteristics. This means that clarification and sludge recirculation cannot be practiced, with the result that bacterial populations are limited by those generated in a single pass oxidation process. If bacterial populations could be increased then the kinetics of the process would improve proportionately. This has been achieved through the use of fixed film reactors. In this approach, bacteria are developed on inert surfaces and exposed to the liquid. As the population increases, so does the number of bacteria exposed to a unit volume of iron bearing wastewater. Both packed column and biodisc treatment processes function in this way, with the biodisc exhibiting a higher efficiency than the packed column.

The process is first order dependent on the ferrous iron concentration and oxygen pressure, and directly proportional to the second power of the hydroxide ion concentration.

Oxidation rates are also strongly temperature dependent at any given pH. See Table 5. This results in residual ferrous iron concentrations that are too high and thus still require pH elevation to 9.5 for complete removal. This is especially the case at low temperatures, i.e. less than 5°C. Further process development is underway.

4.3 Co-precipitation and Adsorption

Co-precipitation is a phenomenon associated with the precipitation of one component from a multi-component system. Component 'A' may be converted to an insoluble form and in so doing may remove a fraction of component 'B' from solution. Under equilibrium conditions component 'B' should have remained in solution.

This phenomenon may be employed advantageously in the precipitation of metal hydroxides. Rapid pH adjustment, resulting in maximum instantaneous hydroxide formation, should result in increased co-precipitation effects, i.e. removal of dissolved materials. This would be in comparison with gradual addition of neutralizing agents and sequential equilibration. The precipitate of calcium sulphate from saturated solutions may also be effective in this regard.

The co-precipitation of radium with barium sulphate is considered in Section 4.8.

Chemical and physical adsorption processes can also be used to advantage in heavy metal removal. Both siliceous and calcareous materials in tailings exhibit affinities for dissolved heavy metal ions. Silica has a zero point of charge in the range of 3 to 4 pH units. At higher pH values it is negatively charged and will adsorb heavy metal ions through electrostatic adsorption. This would also apply to oppositely charged particulates, such as hydroxide particles. This property may be exploited by bringing siliceous gangue into contact with acid waters prior to neutralization. There would probably be a supplementary benefit through improved hydroxide settlement characteristics and sludge consolidation.

Calcareous gangue would act in a manner similar to siliceous materials. It would also effect pH neutralization and hydroxide precipitation. With both classes of materials there would be a finite adsorption capacity which, in practical terms, could not be regenerated. The long term stability of the absorbed contaminants must also be considered. The data available at this time are limited but since the processes are based on ion exchanges it is reasonable to suppose that the absorption is reversible under certain conditions. If large volumes of material are likely to be used in this manner, then stability tests should be run under a variety of field conditions. It should also be recognized that some ions are more easily adsorbed than others. Figure 5 lists data for adsorption on clay of small amounts of metal ions in the presence of large amounts of calcium and sulphate.

4.4 Flocculation/Clarification

In the heavy metals removal process, precipitated hydroxides must be separated efficiently. Gravity sedimentation is universally employed, generally in stilling basins provided by tailings impoundments or lagoons and more rarely in mechanical clarifiers/thickeners. The design and operation of these units is considered in Section 5.

In the majority of cases the hydroxide precipitates are highly flocculated, possess excellent settling characteristics and provide low residual particulates in the supernatant under quiescent conditions.

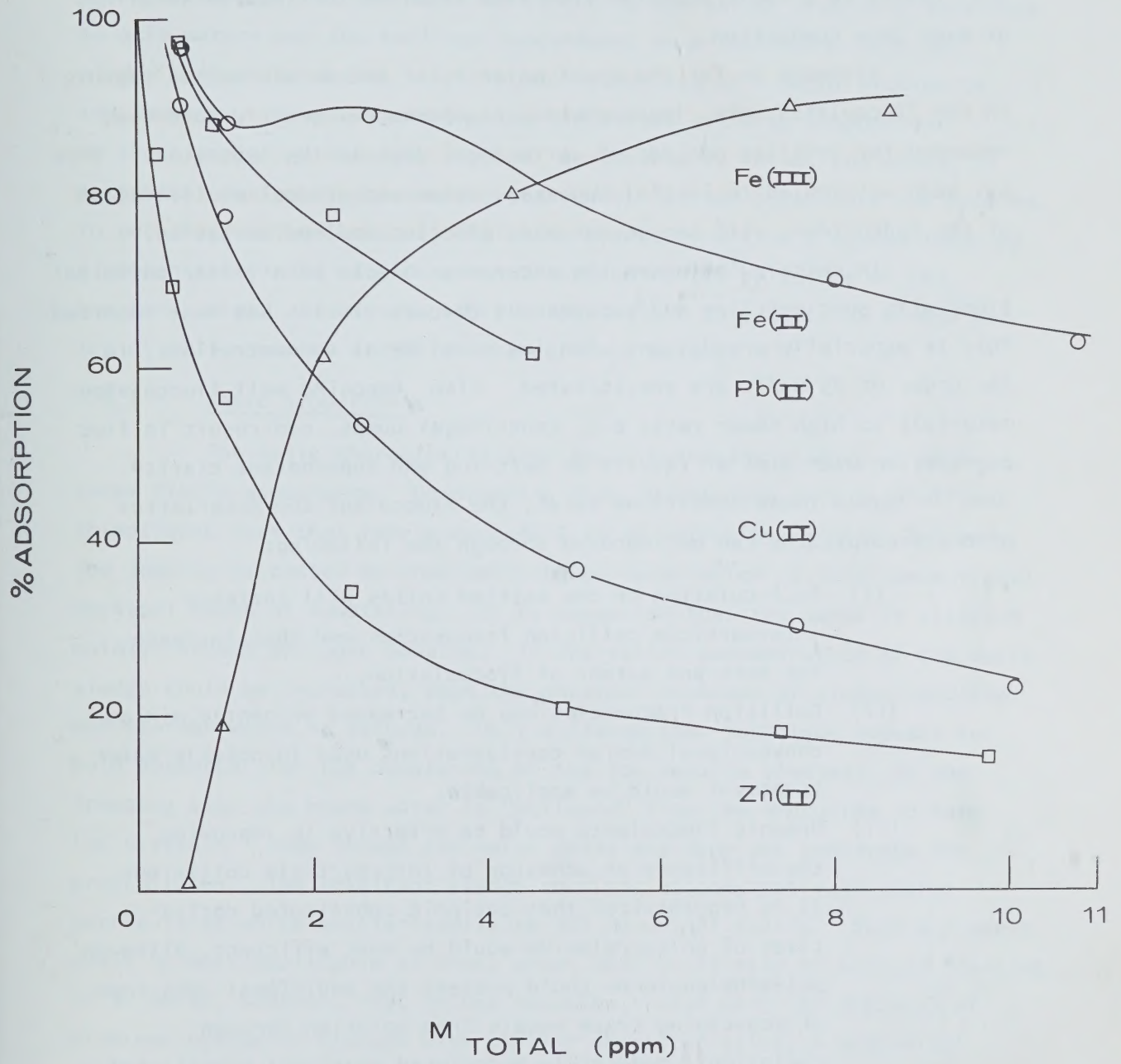


Figure 5. Metal Ion Adsorption on Clay(5)

The basic factors affecting the settling characteristics are the density of the particulates and the occupied volume. A high proportion of sludge will result in a rapid transfer from free settling to hindered settling, or even zone compaction.

Although by far the great majority of the metals settle rapidly in the flocculated mass, improvements in supernatant quality have been recorded for settling periods of up to three days in the laboratory. This has been attributed to initial supersaturation and gradual equilibration of the hydroxides, with consequent precipitation and sedimentation.

In certain instances the occurrence of pin point, near colloidal flocs with poor settling and supernatant characteristics has been reported. This is especially predominant when low total metal concentrations, in the order of 25 mg/l, are precipitated. Also, exposing well flocculated materials to high shear rates e.g. centrifugal pumps, can result in floc degradation with similar results on settling and supernatant clarity.

When these conditions exist, the flocculant characteristics of the precipitates can be improved through the following:

- (i) Recirculation of the settled solids will increase interparticle collision frequencies and thus increase the rate and extent of flocculation.
- (ii) Collision frequencies may be increased by gentle mixing; conventional design considerations used in potable water treatment would be applicable.
- (iii) Organic flocculants would be effective in improving the efficiency of adhesion of interparticle collisions. It is hypothesized that cationic substituted derivatives of polyacrylamide would be most efficient, although polyethyleneimine could possess the additional advantage of scavenging trace metals from solution through chelation. Moderately hydrolysed carboxyl substituted polyactylamides would be applicable if sludge conditioning was the dominant concern.

4.5 Sludge Characteristics

The metal hydroxide sludges produced in a wastewater treatment process represent a highly concentrated stream of contaminated material.

It is imperative that they be efficiently separated from the purified water and disposed of in such a fashion that they will never re-enter the environment. As the majority of mining operations providing treatment to acid waters use the tailings impoundment as a settlement zone and sludge collection device, this is not recognized as a major problem by the industry. However, the problem can assume striking proportions to those employing separate lagoons for treatment. Also, the advent of mechanical equipment treatment systems will further accentuate the problem. To set the problem in focus, the waste sludge flow from a system treating a concentrated acid water may constitute up to 30% by volume of the raw feed water. At one mine this is equivalent to 9 cu ft of sludge per ton of mill capacity or 50% of the tailings volume.

4.5.1 Basic properties

The basic characteristic of metal hydroxide sludges is their loose fluffy appearance. On standing they consolidate into a gelatinous thixotropic mass that rarely exceeds 2 to 3% solids by weight. The very low density is caused by chemically bound water which resists conventional physical means of dewatering. It is suggested that the water is attached mainly through hydrogen bonding. If the solids concentration of the waste sludge could be increased, then the physical problems of sludge handling and storage would be reduced. Only a freeze/thaw technique appears to hold potential for the dewatering of the low density sludges. In the freezing step the bound water is "stripped" from the hydroxide to form ice crystals. When thawed the water melts and does not rehydrate the precipitates. The resultant sludge consists of comparatively coarse particulates which settle readily to 30% by weight solids. Such a process would be most applicable in those areas that could rely on natural freezing. It is noted, however, that in one location, faced with the disposal of aluminum hydroxide sludges with similar characteristics, a mechanical freezing plant was justified for full scale application.

A different approach is employed in the "high density sludge" process developed by the Bethlehem Steel Corporation. In this method, sludge is returned from a final clarifier and blended with lime before being processed through a chemical ferrous oxidation reactor with the

raw acid water. The waste sludge may be thickened to up to 50% solids by weight. The densification process is dependent on $\text{Fe}^{++}/\text{Fe}^{+++}$ ratios and its effectiveness in the presence of other heavy metals has not been established. Although the densification process has been attributed to an ion exchange mechanism, this is in dispute.

Certain extremely acid waters have been reported with sulphate concentrations in the order of 20,000 mg/l. In the presence of calcium ions, introduced through lime or limestone, massive calcium sulphate precipitation will occur. This will dramatically improve the sludge concentration. Sludge densities in excess of 35% solids by weight have been achieved.

Similar benefits are reported when a limestone produced sludge is compared with a straight lime sludge. The combined influence of compaction through unreacted limestone particulates, adsorption on the surface of these particles and the probable formation of dense basic carbonates assist in producing a more consolidated sludge. Heat treatment of precipitated ferrous hydroxide to 80°C has the effect of converting it to the dense magnetite form. This process has not been conducted on a demonstration scale.

4.5.2 Sludge stability

The stability of the sludge with respect to the release of metals is generally indicated by the solubility relationships presented in Figures 1 to 4.

If sludge is stored in surface lagoons it will come into contact with precipitation. Rainwater consists of a weak solution of carbonic acid with typical pH values ranging from 5 to 6 units. In industrialized areas pH values as low as 3.2 to 3.5 have been measured in rainwater and snow and are attributed to the presence of sulphur dioxide. Under equilibrium conditions rainwater will redissolve many heavy metal hydroxide sludges, especially zinc, nickel, ferrous and, to a lesser extent, copper hydroxides. Only ferric iron precipitates may generally be considered to be stable when exposed to the atmosphere. The rate of equilibration is slow and may mask the immediate impact of the process under practical conditions. Nonetheless the data indicate that either (a) the sludge must be converted into a completely stable form; or, (b) affected precipitation must be collected and treated; or, (c) sludge must be disposed of so that

it does not come into contact with atmospheric precipitation; or, (d) the metals contained in the sludges must be recovered.

Table 6 lists data illustrating the problem. A sludge was formed using lime neutralization of water from a Cu-Pb-Zn operation. The treated water was of high quality. The sludge formed was then leached for several days at the pH values shown. Even at pH 5 significant amounts of Cu and Zn are leached.

TABLE 6. DISSOLUTION OF Cu-Pb-Zn SLUDGE (14)
(mg/l)

pH of Leach Solution	4.0	5.0	7.0
Cu	1520	47	0.3
Zn	2330	570	5.3
Fe	4.7	0.2	0.5
Wt. Loss (%)	4.0	1.6	1.2

Various sludge conversions to stable forms have been proposed. They include the formation of stable carbonates and silicates, as well as the use of organic gels. The feasibility of applying these techniques to sludges containing zinc, nickel etc. has not been demonstrated. The treatment of contaminated sludge runoff implies a never ending treatment process. Underground disposal of sludge has not been demonstrated in Canada, although the disposal of ferruginous sludges in abandoned mines has been employed in the United States. Zinc and iron have been recovered from sludges at one location in Germany but no applications are reported in Canada.

4.6 Effluent Stability

The stability of effluents from mining operations should be such that no adverse chemical or biological conversion of the compounds contained in the effluents will occur in the receiving environment. The two basic sources of effluent instability are thiosalts and milling reagents.

4.6.1 Thiosalts

The origin of thiosalts in mining effluents is described in Section 3. The impact of these materials on ambient water quality can be pronounced. Biochemical oxidation of thiosalts with various species of *Thiooxidans* bacteria in the receiving stream results in the formation of sulphuric acid and consequent pH depression. Although the manifestations of this phenomenon have been recognized for a long time, the cause has been established only during the last few years. Recent evaluations indicate that the problem is much more widespread than hitherto suspected. Three basic approaches have been advocated.

(a) Biochemical Oxidation

Bacterial action by a group of autotrophic aerobic bacteria can convert sulphur and its reduced compounds to sulphate. Table 7 indicates several species and their characteristics. Although some recent development work has been undertaken with the bacterium *Thiobacillus thioparus*, most data are available for *Thiobacillus thiooxidans*. The properties of the latter species are considered further.

The optimum conditions for bacterial oxidation are a pH of 3.5 and a temperature of 35°C, in the presence of adequate nutrient supply, oxygen etc. Both completely stirred tank and fixed film reactors are capable of providing the necessary process conditions as described in Section 4.2.2.

A CSTR design has been proposed by Schmidt (7) which incorporates a residence period of six days and is capable of completely oxidizing 1000 mg/l of thiosalts. This would utilize oxygen transfer rates of 5,700 lb/thousand gallons*. If such a process design were to be applied to Canadian mining conditions it would necessitate an outside lagoon or aeration basin which would operate at temperatures of near 0°C in the winter months. The reaction rate of the bacterial oxidation process tends to zero under these conditions and thus would be incapable of providing effluent stabilization in a practicable manner. Field data have confirmed this.

* Unless otherwise noted, gallons referred to in this report are Imperial gallons.

TABLE 7. CHARACTERISTICS OF THE *THIOBACILLI* (7)

Name	Substrates Utilized Other than Thiosulphate	pH Characteristic	Remarks
<i>T. thiooxidans</i>	Sulphur Tetrathionate	Optimum pH 2.0 to 3.0 Range 1.0 to 6.0	Acidophilic; oxidizes thiosulphate rapidly to sulphate.
<i>T. ferrooxidans</i>	Ferrous iron Sulphur Trithionate Tetrathionate	Optimum pH near 3.5 Range 2 to 6	Acidophilic; unable to use sulphur rapidly. Optimum pH for oxidation of sulphur compounds is apparently pH 4 to 5.
<i>T. concretivorus</i>	Similar to <i>T. Thiooxidans</i>	but differs principally in that it can use both ammonia and nitrate.	
<i>T. thioparus</i>	Sulphur Trithionate Tetrathionate Dithionate	Optimum pH near neutrality	Oxidation of substrate causes the medium to become acid.
<i>T. denitrificans</i>	Sulphur Cultivated aerobically this bacterium is indistinguishable from <i>T. thioparus</i> .	Optimum pH near neutrality	Under anaerobic conditions uses nitrate as a terminal respiratory electron acceptor.
<i>T. thiocyanoxidans</i>	Thiocyanate Sulphur Sulphide	Optimum pH near neutrality (pH 6.8 to 7.6)	Produces acid, oxidizes thiosulphate rapidly.
<i>T. novellus</i>	Various organics	Optimum pH near neutrality	Grows slowly, produces acid. Of all the above <i>thiobacilli</i> this is not an obligate autotroph but a facultative autotroph.

Note: All the *thiobacilli* oxidize thiosulphate to sulphate although the precise pathways have not yet been agreed upon.

Fixed film reactors show the same trends in oxidation kinetics but, due to their higher microbial population, exhibit faster and more complete reaction per unit reactor volume. These reactors are at the developmental stage and possess a high potential for in-plant treatment of warm streams containing high concentrations of thiosalts. Recent data have indicated that the oxidation reaction is now always carried through to the sulphate, and that intermediate oxidation products of the polythionate class may be present in significant quantities in process effluents. The causes and significance of this have not been established.

It must be emphasized that, while thiosalts may be found in many mining wastes in sulphidic areas, the concentrations may not be sufficiently high to warrant the installation of specific treatment operations to stabilize them. In many cases, the oxidation provided by normal retention times in a tailings pond may be sufficient without seriously lowering the pH in the system due to thiosalt oxidation. While such removal rates may be reduced during the winter months, it must also be borne in mind that the rate of biological oxidation of the residuals in the receiving streams will also be at a low level and the impact of small thiosalt residuals in such cases may not be significant.

(b) Chemical Oxidation

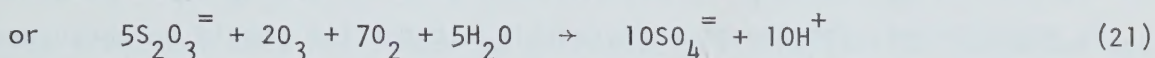
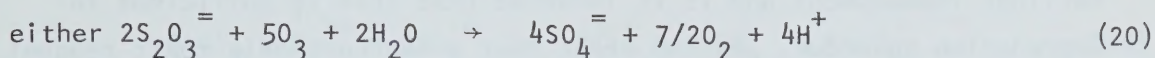
The chemical oxidation of thiosalts may be achieved through the use of chlorine, ozone, sodium chlorate, sodium hypochlorite and hydrogen peroxide. A cost analysis (see Section 5.4.6) indicates that oxidation using chlorine and ozone is most feasible although more expensive than bacterial means. It is worthwhile noting that the oxidation of thiosulphate by atmospheric oxidation is theoretically possible. Some evidence suggests that the catalytic influence of cobalt, nickel salts etc. may be dominant. Unfortunately, this process has not yet been successfully demonstrated.

The following equations have been proposed to describe the oxidation of thiosalts by chlorine and ozone.

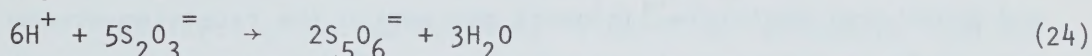
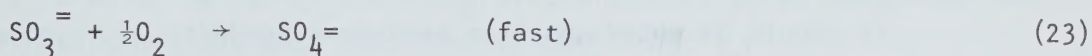
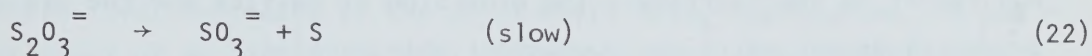
Chlorine:



Ozone:



All chemical and biological oxidation processes for thiosalts will result in acid formation and reductions in the pH of the process liquor. This may be of advantage through the precipitation of elemental sulphur.



Equations (22) and (23) indicate that substantial effluent stabilization may be expected through sulphur precipitation under acid conditions. At pH 3 up to 80% yield as elemental sulphur has been reported. Equation (24) indicates that higher thionates, probably in the form of polythionates and sulphurous acid, are formed. Thus, after complete precipitation of sulphur, residual concentrations of polythionates will exist which may require stabilization.

It may be concluded that both chemical and biological oxidation processes will be influenced by the dissociation of thiosalts under acid conditions.

(c) Addition of Buffer Solutions

The addition of sodium bicarbonate, soda ash, lime or other chemical buffers to the effluent stream provides a possible means of counteracting acid generation by thiosalt oxidation in the receiving stream. Costs, which are summarized in Section 4.1.2, are relatively high and the concept of applying chemical treatment in the receiving stream is not encouraged.

4.7 Process Reagent Stabilization

Although oxidizable organic process reagents are frequently inherently toxic materials (see Section 3.2.1) water quality problems are rarely attributed to this source. The majority of these reagents are subject to biological and/or chemical degradation under the conditions which exist in the tailings impoundment. A 30 day theoretical hydraulic

retention period is generally provided in the pool section of the tailings impoundment and it is inferred that this is sufficient for degradation to occur. In the event that a nondegradable toxic reagent is encountered, the use of a degradable alternative should be encouraged.

In certain instances, the hydrophic characteristics of many reagents may result in the formation of a monomolecular layer on the surface of the tailings pond. This may skim rapidly over the surface and report in the overflow. The provision of baffles and the use of submerged decant weirs can counteract this problem.

It should be noted that the breakdown products of nitrogen and phosphorus containing reagents may enrich the receiving stream and result in symptoms of eutrophication. Phosphates may be precipitated by the use of lime or limestone under alkaline conditions and, where acid conditions are being counteracted by these materials, no additional treatment is generally necessary.

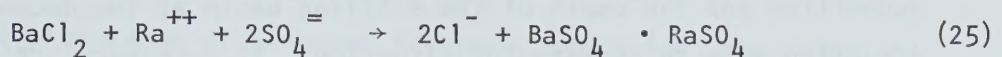
4.8 Radium Removal

Uranium is extracted by leaching processes of two types. Acid leaching is used in the Elliot Lake District, while carbonate leaching is employed at Eldorado because of the high natural carbonate level in the gangue. In most operations the mine water is directed to the mill for removal of dissolved uranium and is subsequently discharged as part of the ion exchange barren solution.

The liquid barren is recombined with the washed filter cake and discharged as mill tailings. Carbonate leach mills require no additional pH adjustment since the net discharge is at pH 8-9. The acid leach tailings are neutralized with lime or a limestone-lime combination and discharged at a pH of 8-10. Since the solid tailings may generate acid, it is desirable to have excess limestone available as a buffering agent. Effluents at pH 8-10 contain very little heavy metals (after sedimentation of hydroxides) and also contain reduced levels of Ra 226 after neutralization, typically from 400-800 pCi/l. The Ra 226 in the barren solution ranges from 3000-4000 pCi/l.

Radium is removed by co-precipitation with BaSO_4 by adding BaCl_2 to the effluent in the presence of excess sulphate (typically 0.01

to 0.1 lb/ton wastewater).



The treatment does not remove Ra effectively if the initial suspended solids are high since barium displaces radium from the solids. The co-precipitate settles slowly and adequate lagoons are needed to reach acceptable levels in the final effluent. The sludge formed may also be unstable if the sulphate level decreases to the point where BaSO_4 redissolves. A low input level of Ra 226 to the BaCl_2 system appears to be beneficial but this result has not been specifically confirmed.

The level of Ra 226 achieved in the final effluents from the various uranium mines varies somewhat because of minor differences in treatment practice and, in particular, because of variations in the sizes of the settling lagoons used. It is difficult, however, to identify specific cause and effect relationships between the effluent levels and the various treatment parameters since there are only three case studies available in Canada at this time (23).

A detailed study and review of the data available was conducted by the Radioactivity Expert Group as a subgroup of the Mining Regulations Task Force (23). This study showed that, using data for filtered samples (3 μ filter), a monthly average level of 10 pCi/l Ra 226 could be achieved using best practicable technology.

4.9 Suspended Solids

The most obvious environmental concern of a mining operation is to eliminate the discharge of tailings into the environment. This can be achieved through the use of an appropriately designed and operated tailings impoundment area. Canadian practice in this regard is considered in Section 5.3.

Although settling rates for the tailings may be measured and used as a process design consideration, this is generally not the controlling factor dictating tailings pond design. Other considerations, such as reagent stabilization requirements, decant design, topography, etc., are usually limiting factors. This results in substantial safety margins for the settling requirements. In those instances where solids carry-over is observed, physical modifications to the side to eliminate

possible short circuiting effects, and to alter the manner of tailings deposition and the depth of the stilling basin at the decant structure, are often able to rectify the situation. If necessary, this may be augmented by improving the settling characteristics through pH adjustment or the addition of flocculating agents.

4.10 Untreated Contaminants Present in Mining Effluents

As we increase our appreciation of the conditions necessary to sustain or improve our environment, we develop a progressively more conservative approach to the allowable discharge of contaminants. In the past, crude observations of parameters such as fish mortality guided the setting of pollution control requirements. The increased awareness of sublethal effects and synergism alert one to the harmful effects of substances that were previously considered completely harmless or were not even known to be present in the effluents from mining operations. Between the identification of a problem and the development of a solution there is always a delay. When the problem is in the field of water pollution abatement from an industrial source, the onus is upon the industry to develop practicable solutions for application to that particular sector or specific location.

The materials listed below are found in mining effluents. They do not occur universally and even when they do occur it is not suggested that they automatically constitute a problem.

The one common characteristic that they possess is that there is no generally accepted means of providing treatment for them as they appear in mining effluents. The alternative approach to that of developing treatment processes is to make in-plant or process modifications to eliminate the contaminant at source.

Typical contaminants are:

- (a) sulphates
- (b) ammonia
- (c) nitrate
- (d) hardness
- (e) chlorides

4.11 Advanced Treatment Processes

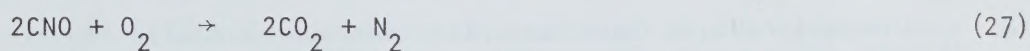
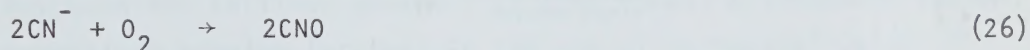
A wide range of advanced treatment processes have potential application in mine wastewater treatment. At the present time, there are economic or engineering problems which limit their use. Their most immediate potential is in environmentally sensitive areas where contaminants currently unaffected by best practicable treatment technology (see above) are causing problems. Table 8 summarizes some prominent advanced treatment methods.

4.12 Cyanide Removal

4.12.1 Base metal mines

Cyanide is used as a depressant for sphalerite and iron sulphide minerals in the flotation of copper and lead minerals.

Free cyanide is unstable under the alkaline oxidizing conditions frequently encountered in the tailings pond. The oxidation reaction may be represented by:



It has been reported that cyanide losses from active mine-mill operations to receiving streams approximate 0.02 to 0.5 per cent of total cyanide mill additions. When this is applied to base metal operations it indicates that free cyanides are adequately stabilized in conventional tailings systems.

Some of the cyanide may complex with heavy metals to form compounds of varying stability, typified by $\text{Cu}(\text{CN})_2^-$, $\text{Fe}(\text{CN})_2^-$, and other metal cyanide complexes. These compounds are not readily susceptible to oxidation in the tailings pond and will report in the decant. One western mill is employing alkaline chlorination for destruction of cyanide prior to discharge but, in this instance, no tailings pond is available.

4.12.2 Gold processing operations

The main difference between wastes containing cyanide from gold processors and base metal processors is the much higher concentrations of cyanide in the former. Values of 500 mg/l are typical to gold barren

*high concentration
of cyanide in
gold.*

TABLE 8. ADVANCED TREATMENT METHODS POTENTIALLY SUITABLE FOR MINE WASTE CONTROL

Process	Description	Present Applications	Present Status	Approximate Costs (2 mgd capacity)	Operating Difficulties and Comments
Electrodialysis	Anion and cation permeable membranes remove respective ions. Membrane transfer driving force is electrical power. Current required depends on concentration.	Desalination Treatment of AMD	650,000 gpd desalination pilot plant 10,000 gpd AMD pilot plant	\$0.50-\$100/10 ³ gal	- cell maintenance problem - hardness causes scale - iron limit of 1 mg/l (influent) - pretreatment needed - concentrate disposal
Reverse Osmosis	Solvent flow across semi-permeable membrane. Pressure on concentrated side in excess of osmotic pressure is driving force.	Desalination Food processing Treatment of AMD	10,000-40,000 gpd pilot plants Field and lab testing	\$0.50-\$1.50/10 ³ gal	- membrane fouling with CaSO₄ - high pressures (600-1500 psi) - concentrate disposal
Ion Exchange	Anion and cation resins replace contaminate ions with H ⁺ , OH ⁻ or others. Regeneration removes these ions into a concentrate stream	Desalination Water softening Treatment AMD Uranium extraction	0.5 Mgd pilot plant (AMD) Well established for many industrial applications	\$0.33-\$1.00/10 ³ gal	- regenerate disposal - hardness - resin selectivity and degradability
Evaporation (Distillation)	Multistage flash evaporation or long tube evaporation. Each evaporates pure water and leaves a concentrated solution.	Desalination Treatment of AMD Petroleum industry	1.0 Mgd desalination plant in use	\$0.60-\$1.50/10 ³ gal	- scaling, corrosion - concentrate disposal
Freezing	Fresh water ice crystals separated, washed and melted. As ice forms, solution becomes concentrated and impurities precipitate.	Desalination Treatment of AMD Food processing	Desalination pilot plant	\$1.40/10 ³ gal	- brine disposal - less power costs than evaporation - metal precipitation could cause scale and corrosion problems
Ion Floatation (Foam Fractionation)	Surface active agent connects dissolved ions to air bubbles. Wastes removed in a froth.	Concentration of trace elements in ocean waste treatment of dilute solutions	Basic research level batch bench scale tests	\$0.71-\$2.16/10 ³ gal	- residual surfactant in product water - hopeful for concentration from dilute solutions - much development work required.
Solvent Extraction	Solute transferred from water to solvent. Driving force is increased solubility in solvent. Solute removed from solvent which is recycled.	Uranium extraction Coke-oven waste treatment Water treatment for organics Treatment of AMD	Widely used in petroleum industry	---	- solvent may pollute effluent - highly developed for organics - solute levels in effluent may still be high - cost depends on solvent recovery
Activated Carbon Adsorption	Adsorption of heavy metals -- Oxidation of other constituents	Iron oxidation in AMD	Other applications under research	low as carbon is catalyst and repeated use is feasible	- intermediate process - not a complete treatment method
Neutrolisis	Combination of Neutralization and Reverse Osmosis by recycling concentrate from R.O. back through neutralization stage.	Treatment of AMD	Two week test run at 10,000 gpd	\$0.50-\$1.50/10 ³ gal	- elimination of CaSO₄ membrane fouling associated with R.O. in flushing cycle - manganese build-up may cause problems - one gallon sludge/100 gallons treated
Ozone Oxidation	Ferrous iron oxidation to ferric followed by neutralization.	Treatment of high ferrous effluents and AMD	Engineering feasibility and economic studies completed	\$0.12-\$0.50/10 ³ gal	- possible netter control, higher throughputs, and lower cost than conventional oxidation - supplement to neutralization process
Precipitate Flotation	Solvent flotation of metal hydroxide precipitates.	None	Laboratory tests - patent pending	unknown	- details unknown due to patent procedures

solutions. The technology to remove or to reduce these levels in the large flows involved to the level of residual cyanide necessary is not yet available in a demonstrated form. Currently, a study is underway by CANMET, Department of Energy, Mines and Resources, in which three methods of cyanide recovery or destruction in gold mill effluents are being investigated. These include the processes of alkaline chlorination, ion exchange and acidification followed by reabsorption of the evolved HCN. This work should be completed by the end of 1975. Subsequent to these studies Environment Canada will propose suitable control levels for incorporation into guidelines and regulations. These proposals will also cover most heavy metals such as Fe, Cu, Zn and Pb since these form stable cyanide complexes which cannot be removed using the normal neutralization-sedimentation process. In all other respects, however, the gold mines are similar to other base-metal mines and can use the common technology.

4.13 Mechanical Treatment Processes

While the most common form of treatment for mining and milling effluents has been the tailings pond or, in some cases, a treatment lagoon, there is currently a growing interest in the use of mechanical waste treatment systems utilizing conventional steps such as pH adjustment, flocculation and clarification. Such plants are now being considered for the treatment of mine waters in the following situations: no mill tailings pond is available; for further treatment of tailings pond effluents and seepage; or, in situations where it is advantageous from a treatment standpoint to treat mine water or surface drainage separately from mill process effluent.

Where a tailings pond is not available, a common method of treating acid metal bearing drainages is to add lime and settle the resulting metal hydroxide precipitates in lagoons. The problem with this approach is the accumulation of low density (1-2 wt %) thixotropic sludge which can rapidly surpass the capacity of the lagoon. One is often faced with the difficult short term problem of sludge disposal or continuously expanding lagoon capacity, which in turn increases the area from which drainage has to be controlled, and the longer term problem of permanent or continuing control of pollution from the lagoon. Once mining operations cease, unless the

post operation problems -
basic pH of the water in the lagoon is maintained, the metal hydroxides will redissolve and the metals will again become a hazard to the environment.

A number of alternatives for the configuration of an appropriate mechanical treatment system are possible, depending on the characteristics of the various effluents, whether it is advantageous to treat the effluents in a common or separate systems, whether a final polishing stage is required, etc. Two of the more probable treatment alternatives are shown in Figures 6 and 7.

A basic consideration of the treatment process depicted in Figure 6 is that, after the primary solids separation has been achieved, the effluent should be stabilized prior to precipitation and sedimentation, where it is necessary to maintain accurate control over process conditions. A polishing step is included which, if necessary, could consist of filtration (i.e., sand) or additional solids removal by lagoon settling.

It is recognized that, in the foreseeable future, tailings disposal as currently practiced will remain the most commonly employed method of disposal. This being the case, there are advantages to be gained by using the tailings impoundment for the primary separation of solids, balancing both contaminated storm flows and quality of effluent to be treated. This approach results in a treatment sequence similar to that indicated in Figure 7.

Once a treatment sequence as described above is contemplated, the economic advantages of minimizing the integrated effluent volume to be treated become apparent. A major advantage of such a scheme is the much reduced surface area exposed to net precipitation that can be designed into the system (smaller tailings ponds, lagoons and contributing areas of surface drainage) which further reduces the final effluent. Furthermore, each operation can be designed and operated so as to fulfill its primary function as efficiently as possible.

The major treatment operations are briefly discussed in the following sections.

4.13.1 Primary solids separation and storage

At many mines, tailings are classified in cyclones or by other means and the coarse 'sands' used for dam construction or mine backfill.

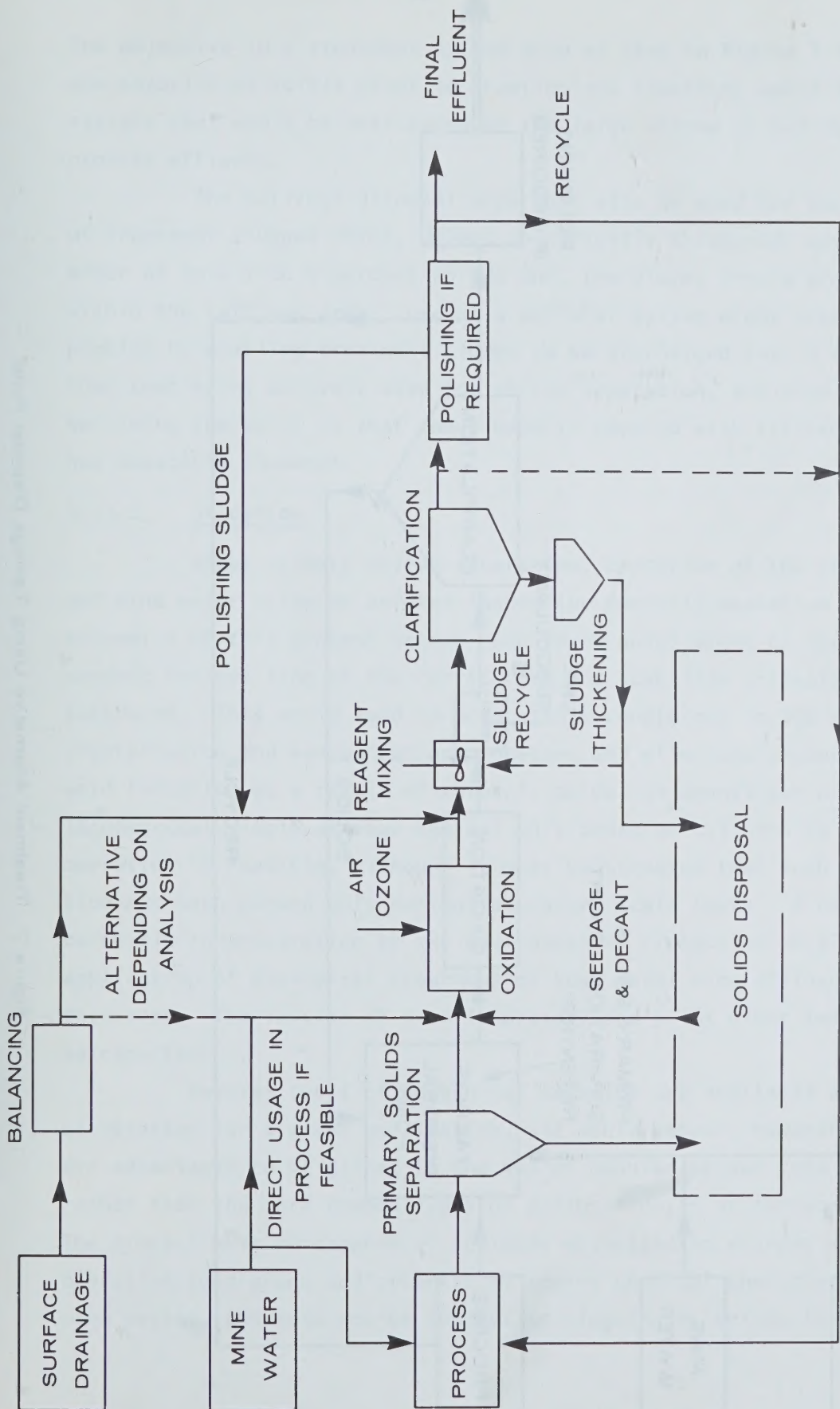


Figure 6 Mine Waste Effluent Treatment Schematic

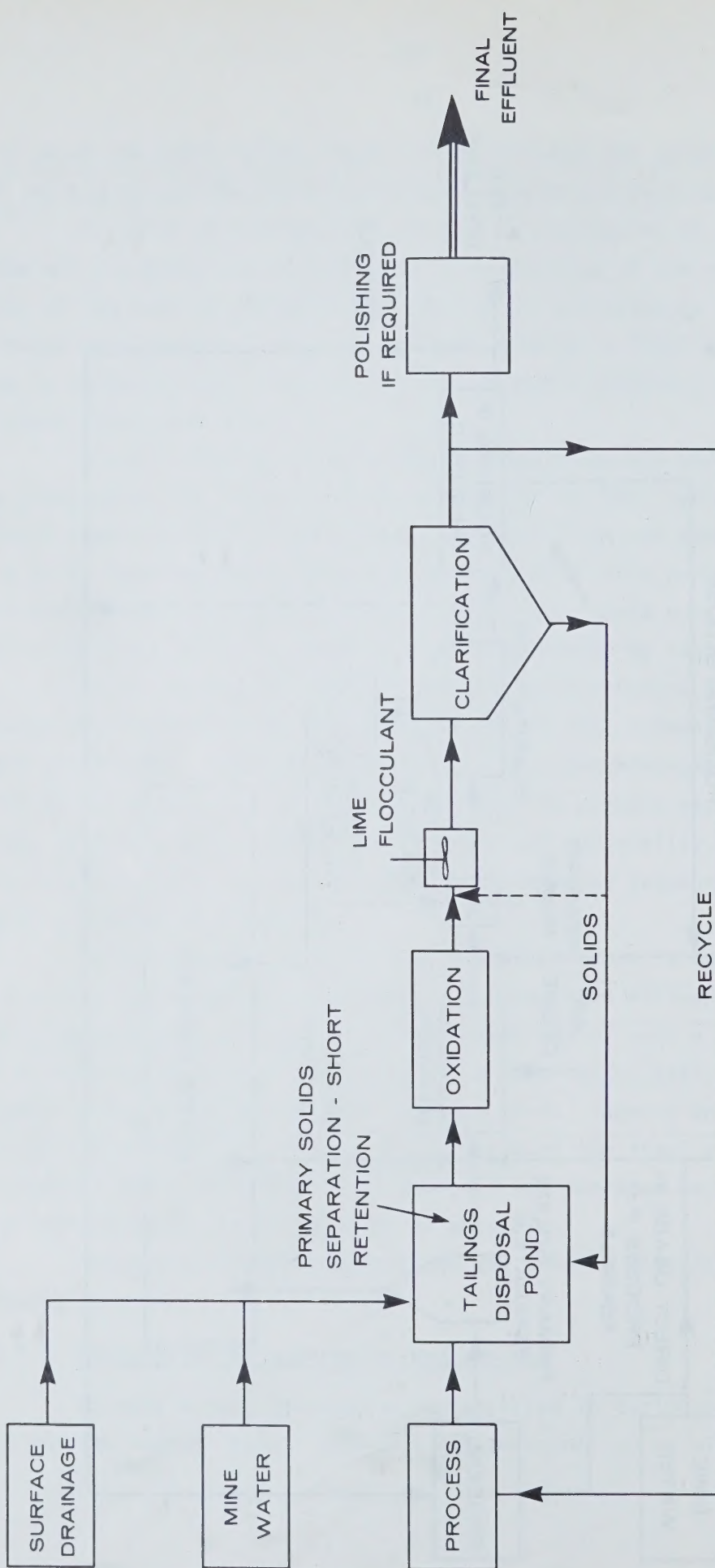


Figure 7. Treatment Alternative Using Tailings Disposal Pond

The objective in a treatment system such as that in Figure 7 is to remove the majority of solids prior to treating the remaining waste stream in systems that would be overloaded by the large volume of solids in the process effluent.

The tailings disposal area must also be used for the disposal of treatment sludges which, unless specifically thickened, may be in the order of only 1 to 2 percent solids and, therefore, create problems within the tailings area. Use of a cellular system might overcome this problem by enabling treatment sludge to be discharged into a cell other than that being actively used for solids separation, and then later switching the cells so that the sludge is covered with tailings once it has dewatered somewhat.

4.13.2 Oxidation

After primary solids separation, oxidation of the process decant and mine water using an aerated lagoon (or possibly ozonation if the economics of this process become more favourable) might be necessary to convert ferrous iron to the ferric form and stabilize thiosalts as sulphates. This would lead to more stable conditions in the subsequent precipitation and sedimentation processes and eliminate uncontrolled acid formation as a result of thiosalt oxidation downstream of treatment. *Thiobacillus* viable at near neutral pH's could be cultured in this operation if feasible, although it must be stressed that such an application has been proven only during laboratory scale tests. A report is currently in preparation by the Environmental Protection Service on the application of biological treatment of base metal mine effluents containing thiosalts. The results of both laboratory and pilot plant tests are to be reported.

Several types of mechanical aeration are available and must be assessed for a given application. It would appear, however, that there are advantages to be gained in the use of tubular or gun type units rather than the more common types of diffuser units or surface aerators. The process must be capable of reliably withstanding extreme winter operation conditions and potentially severe chemical characteristics of mine wastes, and must not be subject to clogging by solids in the system.

Winter operation of the oxidation system is particularly critical but aerated lagoons are successfully employed for the treatment of municipal and organic industrial wastes under conditions equivalent to those in mining regions. An important aspect of the design of such a system is an evaluation of the thermal balance across the system so that the thermal capacity of the waste is employed to prevent freezing. In this respect the oxidation of mining wastes should pose less problems than, for example, municipal wastes, as the temperature of the process effluent is commonly between 10 and 20°C, significantly higher than sewage as it enters a treatment system in winter. The loss of heat in the tailings disposal area must of course be carefully minimized by keeping the retention in winter as low as possible.

An oxidation system, as proposed, would be constructed in two or more cells for ease of maintenance and also because greatly reduced retention times are required in summer. Savings in treatment costs could, therefore, be achieved by reducing the number of units employed in summer as required. Provision of similar aeration facilities with perhaps a six hour retention time may be necessary at mines where clarifiers are being used for precipitate sedimentation, depending upon the amount of ferrous iron present.

4.13.3 Reagent addition

The addition of lime in slurried form, along with the use of a well designed mixing chamber and impeller, involves very little capital or operating costs and can result in very significant savings in reagent costs. Two objectives may be involved in the addition of reagents: pH adjustment, and the improvement of sedimentation characteristics. In the former case, it is essential that reagent additions are regulated by a pH sensor control system, which is common practice in many similar industrial applications. The addition of sedimentation aids is similarly critical and must be quantitatively controlled. Contact chambers and, where applicable, flocculation facilities are essential components of a well designed reagent mixing system. Aeration is often found to be an economic advantage at the reagent mixing stage as it expels free CO₂ from the water and thus reduces its acidity and thereby the addition of

alkali reagents. This is particularly true where limestone or other carbonates are being used as reagents.

4.13.4 Clarification

The provision of a well designed and operated system of clarifiers should ensure that optimum sedimentation conditions for the removal of heavy metal precipitates and other suspended solids are consistently maintained. For the removal of metal hydroxides it is likely that relatively low overflow rates in the range of 300 to 1000 gpd/ft² will be necessary, due to the generally light nature of the flocs.

In practice, the clarifier may be physically combined with either a flocculator, as a clarifier/flocculator, or with a sludge thickening capability, as a clarifier/thickener, but this is a matter for individual economic evaluation in a given application.

4.13.5 Polishing

As previously mentioned, the need for advanced treatment polishing techniques subsequent to the suggested treatment sequence is not foreseen in the vast majority of cases. However, it may be necessary in some cases to further polish the clarifier overflow to produce the desired effluent quality. The results obtained of polishing by means of both sand filtration and lagoon settling in a pilot plant project are shown in Section 4.14.3.

4.13.6 Other treatment variations

Neutralization of mine water and contaminated surface drainage that are high in iron results in waste sludges that commonly have a very low solids ratio, are difficult to thicken by standard processes, and therefore constitute a handling and disposal problem. This situation was encountered in many instances of acid mine drainage treatment in the Appalachian region of the U.S. and elsewhere.

A modification of the neutralization process (called the High Density Sludge Process) which greatly reduces these difficulties has been developed by Kostenbader and Haines of Bethlehem Steel Corporation (8). In essence, it involves drawing off the hydroxide sludge formed by lime neutralization of acid mine drainage, mixing it with more lime and

recycling the mixture back into the primary treatment stream. This has resulted in increased sludge densities (in the order of 30 per cent for sludge recycle ratios above 15:1) and requires only the addition of a second reaction tank and a slurry return system to modify a standard lime neutralization system. The conventional and High Density Sludge neutralization techniques are shown in Figure 8.

The high density sludge is more amenable to further dewatering by centrifuging or filtering than the conventional neutralization sludge, which means that it is technically feasible to dry the sludge to a cake for final disposal if required. Further mention of this process is made in Section 4.14 where its use in the New Brunswick acid mine water pilot plant project is discussed.

4.14 Pilot Plant Treatment Project

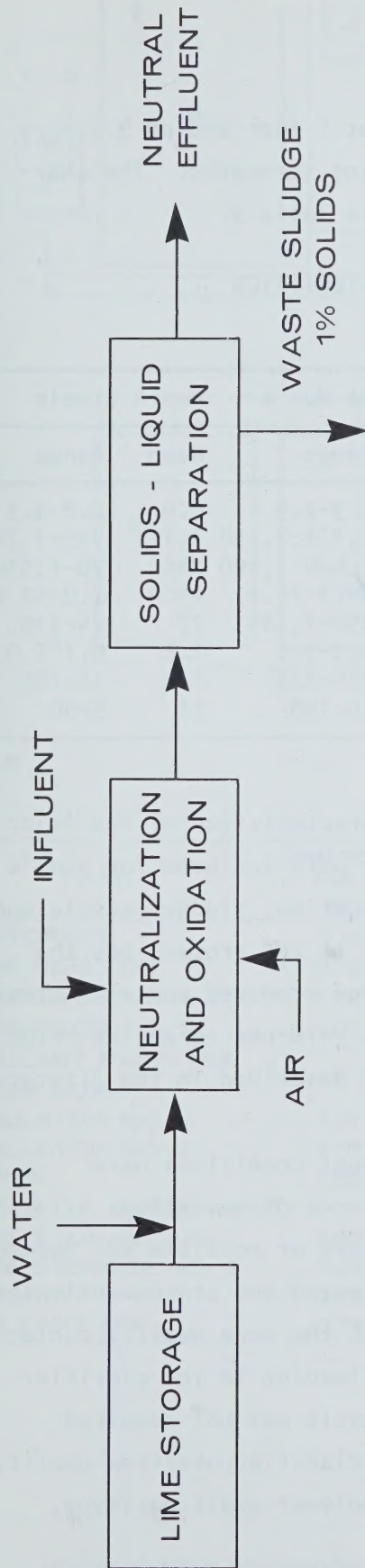
A mine water treatment pilot plant was operated from March, 1973 to October, 1974 at Brunswick Mining and Smelting Company as a joint project between the mining company, the Province of New Brunswick and the federal government. A detailed report of the project was released in May 1975 (9) as well as earlier reports (10, 11) throughout the project.

4.14.1 Objectives

The objectives of the pilot plant program were:

- 1) to determine the best attainable effluent levels for copper, iron, lead and zinc using conventional precipitation and sedimentation techniques;
- 2) to define the optimum performance characteristics of the major components in the treatment sequence, particularly iron oxidation, metal precipitation and sedimentation;
- 3) to determine the characteristics and assess means of handling and dewatering the sludges produced;
- 4) to investigate sludge densification by the High Density Sludge process and establish its effect on effluent metal levels; and,
- 5) to evaluate various effluent polishing techniques subsequent to conventional treatment.

CONVENTIONAL PROCESS



HIGH - DENSITY SLUDGE PROCESS

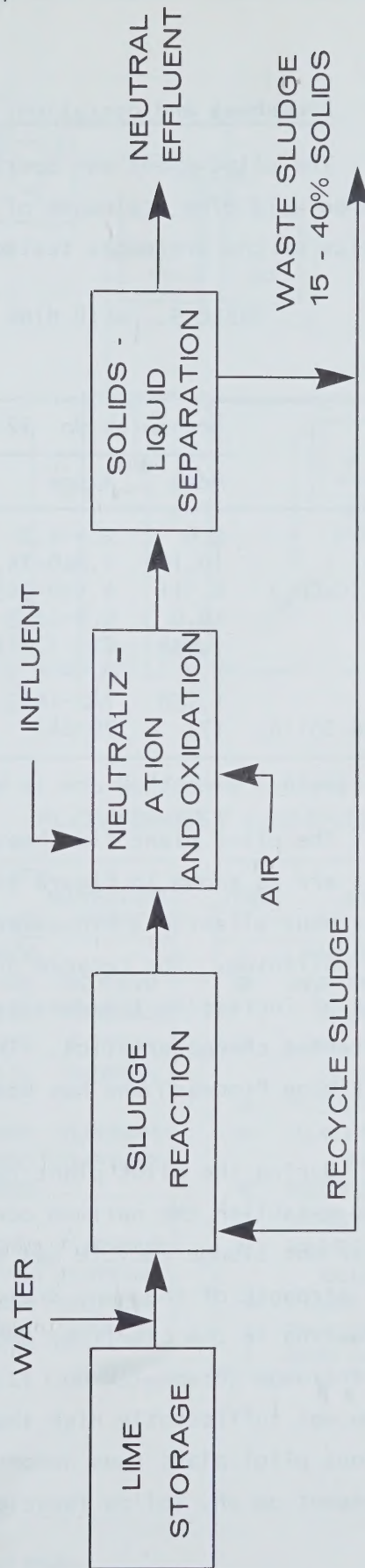


Figure 8. Conventional and High Density Sludge Treatment Systems

4.14.2 Flowsheet and operations

The pilot plant was operated mostly at 5 lgpm and pH 9.5 using three acid mine drainages of widely varying strengths. The characteristics of the drainages tested are given in Table 9.

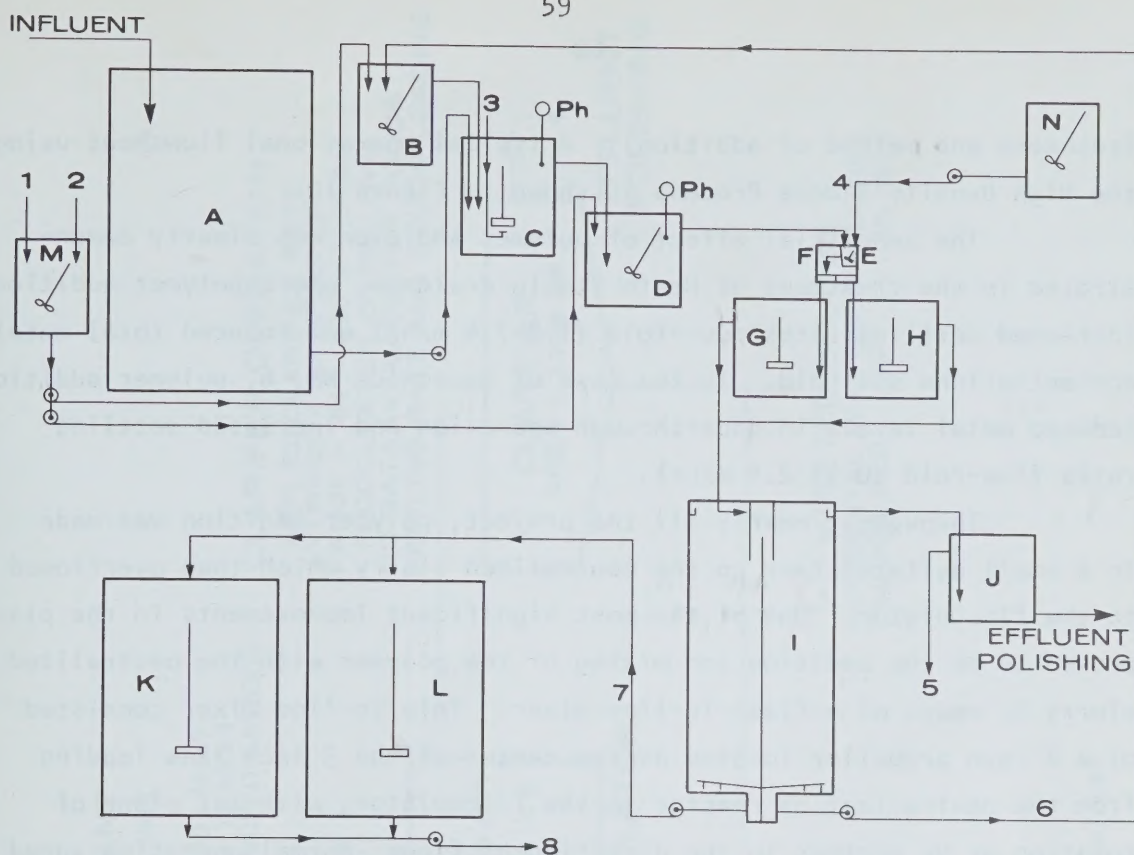
TABLE 9. ACID MINE WATER CHARACTERISTICS

Parameter*	Brunswick No. 12		Brunswick No. 6		Heath Steele	
	Mean	Range	Mean	Range	Mean	Range
pH	2.6	2.4-3.2	2.7	2.3-2.9	3.0	2.8-3.3
Sulphate	10,100	1,860-14,892	4,454	2,354-7,290	1,121	729-1,790
Acidity (CaCO_3)	6,511	4,550-9650	4,219	2,600-7,000	746	70-1,530
Copper	10.0	4.8-22.3	47.2	24.3-76.0	19.4	1.0-52.0
Iron	1,534	8.5-3,211	718	350-1,380	77	24-230
Lead	3.9	0.9-10.3	1.2	0.3-3.2	1.3	0.1-5.0
Zinc	1,158	142-1615	538	390-723	114	18-185
Suspended Solids	172	70-645	65	10-190	31	5-90

* All parameters except pH are in mg/l.

The pilot plant flowsheet and the characteristics of the major equipment are as shown in Figure 9. Provisions were included for single or two stage neutralization, flocculation, clarification, sludge recycle and effluent polishing. The recycle of sludge back to the process has the advantage of increasing the density of the sludge produced and of improving its dewatering characteristics. This system is referred to as the "High Density Sludge Process" and has been previously described in the literatures (8).

During the pilot plant runs, operational conditions were varied to establish the optimum conditions and mode of operation. Whether or not sludge recycle was either necessary or possible was dependent upon the strength of the mine drainage being treated and consequently the solids loading in the clarifier underflow. With the more heavily contaminated drainage (Brunswick No. 12) the solids loading in the clarifier underflow was sufficiently high that sludge recycle was not required. The various pilot plant runs demonstrated that clarifier overflow quality was dependent on pH, solids recycle ratio and polymer addition (type,



FLWSHEET LEGEND

M	TANK	DIA.		CAPACITY		IMPELLER TYPE
		M	HT. M	LITERS	USG	
	AMD STORAGE	2.13	3.35	10598	2800	
	SLUDGE REACTOR	0.76	1.22	473	125	SCREW
	AMD REACTOR NO. 1	1.07	1.37	1098	290	TURBINE
	AMD REACTOR NO. 2	1.07	1.22	946	250	SCREW
	FLOCCULANT RAPID MIX			5.8	1.5	SCREW
	SPLITTER BOX					
	FLOCCULATOR NO. 1	1.07	1.22	946	250	TURBINE
	FLOCCULATOR NO. 2	1.07	1.22	946	250	TURBINE
	CLARIFIER	1.52	3.05	5678	1500	
	CLARIFIER O.F. SURGE TANK					
	SLUDGE STORAGE NO. 1	1.83	2.44	6623	1750	TURBINE
	SLUDGE STORAGE NO. 2	1.83	2.44	6623	1750	TURBINE
	LIME SLURRY	0.76	1.22	473	125	SCREW
	FLOCCULANT MIX	0.76	1.22	473	125	SCREW

ITEM	PROCESS FLOW
1	LIME SLURRY - 18% SOLIDS
2	WATER - LIME DENSITY CONTROL
3	LOW PRESSURE AIR (3 PSIG)
4	FLOCCULANT
5	CLARIFIER OVERFLOW - TO WASTE
6	RECYCLE SLUDGE
7	WASTE SLUDGE
8	WASTE SLUDGE TO SLUDGE HAND'G
⑨	PUMP
○	Ph-CONTINUOUS MONITOR AND AUTOMATIC CONTROL

Figure 9 Mine Wastewater Treatment Pilot Plant

freshness and method of addition). A typical operational flowsheet using the High Density Sludge Process is shown in Figure 10.

The beneficial effect of polymer addition was clearly demonstrated in the treatment of Heath Steele drainage, where polymer additions increased settling rates four-fold (1.8-7.4 m/hr) and reduced total metal concentrations six-fold. In the case of Brunswick No. 6, polymer addition reduced metal levels in once-through operation and increased settling rates five-fold (0.45-2.4 m/hr).

Throughout nearly all the project, polymer addition was made in a small agitated tank to the neutralized slurry which then overflowed to the flocculator. One of the most significant improvements in the plant proved to be the addition and mixing of the polymer with the neutralized slurry by means of a flash in-line mixer. This in-line mixer consisted of a 2 inch propeller located at the center of the 3 inch line leading from the neutralization reactor to the flocculator, with its plane of rotation at 90 degrees to the direction of flow. Normal operating speed and retention times were 1200 rpm and approximately 1 second, respectively.

Sludge recycling and flocculant addition resulted in significant increases in the sludge solids content of the clarifier underflow, particularly in the cases of the weaker mine drainages. A summary of the effects of these two factors is given in Table 10.

TABLE 10. EFFECT OF SLUDGE RECYCLE AND FLOCCULANT

Operating Conditions	Brunswick No. 12		Brunswick No. 6		Heath Steele	
	Recycle Ratio	% Solids Clar. U/Flow	Recycle Ratio	% Solids Clar. U/Flow	Recycle Ratio	% Solids Clar. U/Flow
No recycle, no polymer				3.4		1.4
No recycle, polymer		13.0		7.2		3.1
Recycle no, polymer					5.3	6.8
					14.3	14.0
Recycle, polymer	3.0	25.2	1.4	14.2	40.0	14.7
	2.5	18			18.0	22.8

Recycle Ratio - lb solids recycled/lb solids wasted.

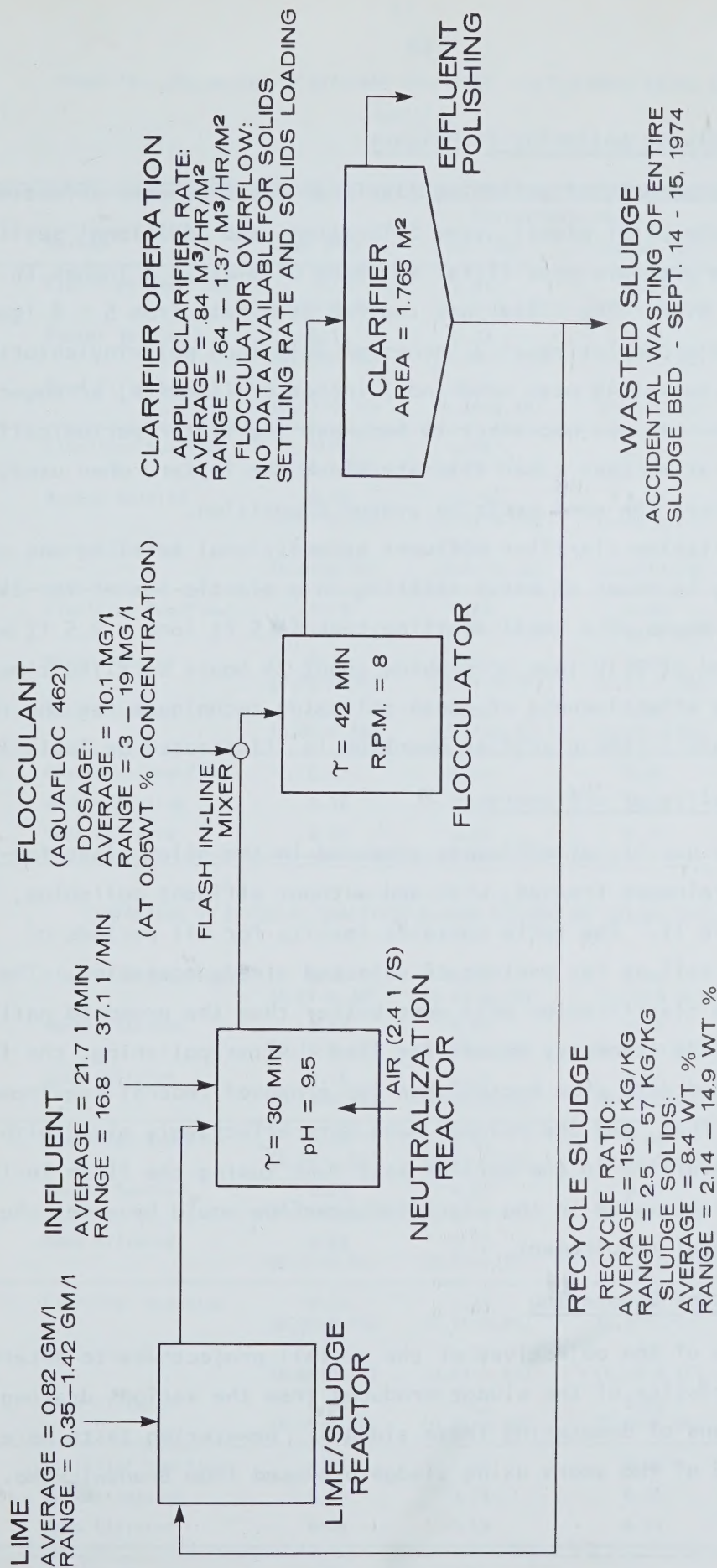


Figure 10 Heath Steele Drainage Operational Flowsheet

4.14.3 Effluent polishing techniques

Two methods of polishing clarifier overflow were effectively employed in the pilot plant: sand filtration, and additional settling.

The pressure sand filter was made of plastic 8 inches in diameter and 3.8 feet high. The filter was top fed at a rate from 5 - 8 l_gpm/ft² through a media consisting of 3 inches of 0.12 inch polyvinylchloride cubes, 12 inches of 40 mesh sand and 7 inches of ilmenite, arranged from top to bottom. It was necessary to backwash the filter periodically with fresh water rather than clear filtrate since the latter, when used, tended to cement the sand media by gypsum deposition.

Polishing clarifier effluent by additional settling was carried out initially by means of batch settling in a plastic bucket for 24 hours and later by means of a small settling tank (4.5 ft long x 2.5 ft wide x 2 ft deep) fed at 0.10 l_gpm to provide about 24 hours settling time.

The effectiveness of these polishing techniques for the reduction of total metals in the clarifier overflow is illustrated by Table 11.

4.14.4 Quality of effluents

The quality of effluents produced in the pilot plant for all three mine drainages treated, with and without effluent polishing, is shown in Table 11. The table contains results for all periods of operation as well as for periods of selected steady operation. The results obtained from clarification only were better than the proposed national effluent quality standards except for lead. After polishing, the lead values attained were also better than the proposed federal requirements. It is likely that, had the polymer been more effectively mixed with the neutralized slurries in the earlier test runs (using the flash in-line mixer), the lead value in the clarifier overflow would have met the proposed federal requirement.

4.14.5 Sludge Dewatering

One of the objectives of the overall project was to determine the characteristics of the sludge produced from the various drainages and to assess means of dewatering these sludges. Dewatering tests were run over a period of two weeks using sludge produced from Brunswick No. 12

TABLE 11. COMPARISON OF EFFLUENT QUALITIES - EXTRACTABLE METAL LEVELS
(mg/l)

Drainage	Stream	Extractable Metal			
		Lead (Pb)	Zinc (Zn)	Copper (Cu)	Iron (Fe)
Brunswick No. 12	Clarifier Overflow	0.18 (0.05-0.39)	0.41 (0.13-0.87)	0.04 (0.02-0.10)	0.30 (0.14-0.65)
	Bucket Settled	0.18 (0.08-0.25)	0.23 (0.20-0.25)	0.03 (0.01-0.05)	0.16 (0.08-0.29)
	Sand Filtered	0.12 (0.07-0.15)	0.26 (0.14-0.38)	0.03 (0.02-0.04)	0.23 (0.09-0.63)
Brunswick No. 6 (recycle flocculant)	Clarifier Overflow	0.35 (0.05-0.62)	0.52 (0.07-1.42)	0.06 (0.03-0.19)	0.54 (0.12-2.51)
	Bucket Settled	0.26 (0.01-0.50)	0.26 (0.03-0.60)	0.03 (0.02-0.07)	0.20 (0.04-0.41)
	Sand Filtered	0.31 (0.01-0.50)	0.28 (0.03-0.58)	0.03 (0.02-0.04)	0.11 (0.02-0.20)
Heath Steele	Clarifier Overflow	0.13 (0.05-0.62)	0.64 (0.14-1.45)	0.10 (0.01-0.30)	0.47 (0.09-1.40)
	Basin Settled	0.11 (0.05-0.36)	0.29 (0.01-0.74)	0.05 (0.01-0.30)	0.26 (0.01-0.61)
	Sand Filtered	0.10 (0.05-0.36)	0.21 (0.01-0.75)	0.04 (0.01-0.30)	0.22 (0.01-0.89)
Overall Mean Levels Attained	Clarifier Overflow	0.21	0.53	0.06	0.43
	Bucket Settled	0.16	0.28	0.04	0.23
	Sand Filtered	0.15	0.23	0.04	0.19

COMPARISON OF EFFLUENT QUALITIES DURING PERIODS OF STEADY OPERATION

Brunswick No. 12 Drainage December 12 - December 31	Clarifier Overflow	0.18 (0.01-0.35)	0.33 (0.13-0.52)	0.04 (0.03-0.06)	0.19 (0.10-0.26)
	Bucket Settled	0.21 (0.16-0.25)	0.29 (0.28-0.30)	0.04 (0.03-0.04)	0.18 (0.15-0.21)
	Sand Filtered	0.15 (0.14-0.15)	0.39 (0.38-0.39)	0.03 (0.03-0.03)	0.20 (0.14-0.26)
Brunswick No. 6 October 24 - October 31	Clarifier Overflow	0.44 (0.25-0.62)	0.45 (0.27-0.69)	0.05 (0.03-0.07)	0.42 (0.14-0.45)
	Bucket Settled	0.29 (0.01-0.50)	0.22 (0.03-0.60)	0.03 (0.02-0.03)	0.17 (0.04-0.29)
	Sand Filtered	0.29 (0.17-0.42)	0.15 (0.03-0.28)	0.03 (0.02-0.03)	0.13 (0.11-0.17)
Heath Steele September 13 - September 22	Clarifier Overflow	0.15 (0.09-0.25)	0.35 (0.14-0.90)	0.06 (0.03-0.11)	0.26 (0.09-0.60)
	Basin Settled	0.11 (0.05-0.18)	0.22 (0.13-0.40)	0.07 (0.02-0.15)	0.24 (0.10-0.30)
	Sand Filtered	0.08 (0.05-0.22)	0.12 (0.03-0.18)	0.03 (0.02-0.04)	0.14 (0.08-0.23)
Mean Levels Attained During Steady Operation	Clarifier Overflow	0.25	0.37	0.05	0.28
	Bucket Settled	0.14	0.23	0.05	0.21
	Sand Filtered	0.09	0.15	0.03	0.14
Proposed National Guideline	Monthly Average	0.2	0.5	0.3	1.0
	Grab Sample	0.4	1.0	0.6	2.0

drainage. The following four pilot scale dewatering units were tested: (1) continuous scroll solid bowl centrifuge; (2) vertical basket centrifuge; (3) rotary vacuum drain filter; and, (4) plate and frame pressure filter.

The properties of the sludge tested were:

<u>Parameter</u>	<u>Value</u>
Solids Content	7%
Bulk Density (at 7% solids)	1.044 gm/cm ³ (65.2 lb/ft ³)
Dry Solids Density	3.26 gm/cm ³ (204.6 lb/ft ³)
Zeta Potential	0
Mean Particle Size	4.9 μ m (0.00019 in)
Specific Resistance	8.4×10^7 sec ² /gm (1.85×10^5 sec ² /lb)
Capillary Section Time	20 sec
pH	10
Conductivity	2.81 millimhos
Sludge Viscosity	4.7 centipoise
Filtrate Viscosity	1.03 centipoise
Average Metal Analysis (% Wt)	0.1% Pb, 6.5% Zn, 0.07% Cu, 6.5% Fe

Cake solids of 25% and suspended solids recoveries of 95% were attained on all four pilot scale dewatering units. The use of a basket centrifuge may not be practical due to the very short cycle times resulting from the high solids concentration (~7%) of the feed sludge. The addition of cationic polymer was necessary to achieve recoveries in excess of 75% with the centrifuges, but was not required with either the vacuum filter or the filter press. The filter press, which produced a filter cake of 38 to 45% and a filtrate quality of 0 to 10 mg/l suspended solids, proved to be the best dewatering unit with respect to cake solids and effluent quality. However, the vacuum filter would require the lowest operating cost of the units tested.

A more detailed report on dewatering sludges produced from the treatment of acid mine drainages has been presented by Dr. B.P. LeClair and H. Campbell (12).

4.14.6 Principle findings and conclusions

The principal findings from the pilot plant project were as follows:

1) The following metal levels were attained as clarifier overflow concentrations, on an averaged basis, for the various drainages treated during periods of steady operation:

	<u>Extractable Metal Levels</u> (mg/l)	<u>Dissolved Metal Levels</u> (mg/l)
Lead	0.25	0.24
Zinc	0.37	0.26
Copper	0.05	0.04
Total Iron	0.28	0.22

2) Polishing of the clarifier overflow by sand filtration and bucket settling further reduced the above extractable metal levels. Levels attained on an averaged basis were:

	<u>Extractable Metal Levels</u> (mg/l)
Lead	0.12
Zinc	0.19
Copper	0.04
Total Iron	0.17

Effluent polishing also reduced the variation in final effluent quality.

3) The initial strength of the drainage had little effect on the final effluent quality but greatly influenced the volume and density of sludge produced, and also the quantity of neutralizing reagent required. For all three drainages tested, the lime demand was approximately equal to the acidity, expressed as mg/l CaCO_3 .

4) The most critical process in the pilot plant operations, relative to obtaining low clarifier overflow metal levels, was complete and thorough mixing of the polymer with the neutralized slurry. The need to prepare flocculant in small daily batches to avoid the deleterious effect of polymer aging was also identified. The optimized method of polymer addition developed during the program was to add the polymer by means of an in-line mixer which operated at high speed with a short contact time.

- 5) Dense clarifier underflow sludges (>15% by wt) were obtained without sludge recycle in the case of the two stronger drainages, with a maximum of about 40% by weight attainable with the strongest drainage. Sludge recycle (High Density Sludge Process) proved to be an effective technique for increasing the density of the sludge resulting from the treatment of the weakest drainage, for which an increase in sludge density from 2% to about 15% by weight was effected, with a solids recycle ratio of approximately 15 kg of solids recycled per kg of solids produced.
- 6) There was found to be no performance advantages in using two stage lime neutralization compared to single stage lime neutralization since the sensitivity of the process was a function of solid-liquid separation and not pH, provided that the pH was maintained within one pH unit of the optimum.
- 7) The most critical operating difficulties identified were as follows:
 - a. Gypsum scaling in the neutralization reactor. The rate of scale build-up was very rapid with the strongest drainage. Physical chipping and removal of the scale was found to be the only way to control its build-up. For the weaker Heath Steele drainage, however, the rate of scale formation was hardly significant.
 - b. Withdrawal of sludge from the clarifier if the sludge density exceeded about 20% by weight. In such cases the sludge would bridge, rather than flow, into the recycle pump, resulting in damage to the pump.
- 8) In general terms, it was found that there were acceptable ranges for most operating parameters rather than sharply defined optimum operating points. Variations in retention times in the various reactors and flocculators, and variations in pH of approximately one unit about the "optimum" of 9.5 did not affect performance.
- 9) The sand filters used as a means of effectively polishing the clarifier overflow were prone to coating and subsequent binding of the sand particles by precipitated gypsum unless regularly backwashed with fresh water. This polishing technique requires further development to enable filtrate to be used for backwashing purposes.

10) Quiescent settling was shown to be an effective method of further reducing metal values in clarifier overflow and reducing the variability in these levels.

11) Sludge dewatering tests performed on both bench and pilot scale units, with the sludge produced in treating the strongest drainage, indicated that the sludge could be effectively dewatered by vacuum filtration or pressure filtration, while centrifugation proved to be less effective. Pressure filtration produced a filter cake of 38% to 45% solids and a filtrate quality of 0 to 10 mg/l suspended solids.

12) The median lethal times (LT_{50}) for salmonid test fish used in bioassay tests on drainages treated in the plant were consistently in excess of 96 hours (the maximum test period), provided that the pH of the treated drainage was adjusted to near neutral values before the tests were conducted.

5. CANADIAN PRACTICE OF EFFLUENT CONTROL

In preceeding sections of this report the source and character of contaminants derived from mining operations, as well as basic treatment processes, have been considered. The aim of this section is to describe Canadian practice in the field of water pollution abatement in the mining industry.

Basically, three kinds of waste flows may be encountered in mine-mill operations, namely, acid mine drainage, mill process effluent and contaminated surface drainage, including tailings pond seepage. By far the most common means of treating these wastes is to discharge them into a tailings pond in which the suspended solids (tailings) are settled out, the heavy metal ions in the water may be precipitated and settled out, and the solids are retained in perpetuity. In locations where large tonnages of solids are involved, such as in the waste effluent from a mill, a tailings pond is employed for the removal and containment of the solids and treatment of the accompanying water. However, at mines where no mill exists, mine water and contaminated surface drainage may be chemically treated in lagoons or handled in a mechanically operated treatment facility. At the present time little use is being made of the latter method, although the construction of some large capacity plants is planned shortly.

5.1 Tailings Ponds

The mining industry is perhaps unique in the quantity of waste solids generated in the recovery of its products from ores. At many mines, particularly underground mines, the mill tailings form the largest portion of the solid wastes produced. The disposal of these tailings exerts a major influence on the design of a mine-mill complex.

Originally, and until not too long ago, the sole purpose of a tailings impoundment area was to prevent the discharge of substantial amounts of solids into receiving water courses. This, of course, is still a primary requirement of a tailings pond, but over the past few years the situation has changed greatly as concern over the deterioration of the environment has increased, along with our knowledge of the damaging effect on aquatic and human life of various soluble constituents present

in wastewaters. In many instances, the tailings pond is now called upon to provide a large degree of chemical treatment to the liquid wastes within the pond. A typical pond may be required to perform some or all of the following functions:

- 1) removal of tailings solids by sedimentation;
- 2) acid neutralization;
- 3) formation of heavy metal precipitates (hydroxides);
- 4) sedimentation of metal precipitates;
- 5) perpetual retention of settled tailings and precipitates;
- 6) stabilization of oxidizable constituents, e.g. thiosalts to sulphates and flotation reagent residuals;
- 7) balancing action for fluctuations in influent quality and quantity;
- 8) storm water storage and flow balancing; and,
- 9) water treatment facility for recycling water to the mill.

Undoubtedly tailings ponds have been virtually universally accepted as the primary means of treatment due to their ability to effectively perform most of the above listed functions at a relatively low cost.

Counteracting the treatment efficiency often achieved in tailings ponds is the lack of control that can be exercised over them, particularly in view of conflicting requirements. For example, sedimentation requires quiescent conditions whereas oxidation of soluble constituents is improved by turbulence.

5.1.1 Advantages and disadvantages

The relative advantages and disadvantages of tailings ponds are:

Advantages:

- a) performs large number of treatment functions at a low cost;
- b) can achieve high treatment efficiencies and produces an acceptable effluent quality in many situations;
- c) often the only practical means of solids disposal;
- d) large retention time has a balancing effect on effluent quality;
- e) large surface area aids oxidation and evaporation;
- f) can often be constructed using mining equipment and materials;

- g) little operating expertise normally required; and,
- h) commonly used treatment method familiar to the industry.

Disadvantages:

- a) lacks responsive means of control, difficult to optimize the large number of processes performed;
- b) covers large surface area and contributes high volume from precipitation to the overall water balance. (This, in cases where recycling of tailings water is practised, is not necessarily a disadvantage. Precipitation will have a dilution effect on the quality of the reclaim water where high concentrations of certain dissolved materials would be detrimental to the process. The large quantity of precipitation will also reduce raw water requirements);
- c) creates a potentially severe rehabilitation problem if tailings are acid forming;
- d) often difficult to isolate from contributing drainage areas - storm water influences retention time;
- e) subject to climatic variations, particularly thermal skimming and seasonal variation in bio-oxidation efficiency;
- f) often difficult to ensure good flow distribution;
- g) requires careful control of seepage through dams;
- h) installation is expensive in many situations due to high cost of retaining structures and need for good engineering design to prevent problems such as liquefaction and piping, followed perhaps by catastrophic failure of the embankment; and,
- i) dust problems are often observed unless surface stabilization is achieved by revegetation or by the use of chemical binders or rock covers.

5.1.2 Location of tailings and waste rock dumps

The following are to be considered as statements of principle concerning the disposal of tailings and waste rock from mining operations.

- 1) Confined disposal in appropriately engineered structures is preferred under all circumstances.

- 2) Unconfined disposal in lakes, swamps, marshes and at sea should be considered on a case by case basis and only when it is demonstrated that confined disposal is impractical and potentially more hazardous than unconfined disposal. If approval is obtained for subaqueous disposal, deep water disposal practices are to be employed whenever practical.
- 3) Disposal of tailings or waste rock into a river is to be prohibited.

5.2 Other Disposal Methods

It is considered that non-land disposal practices will generally have a far greater potential for detrimental impact on the environment. When other than land disposal is proposed, a detailed case by case assessment of each application would be made.

The following points should be considered by the applicant in order to minimize any adverse environmental impact when considering non-land disposal. There is no implication that, even if an applicant were to employ approved procedures for minimizing the impact, the application would be either acceptable or accepted.

5.2.1 Shallow water disposal

The disposal of tailings and waste rock into marshes, swamps, shallow lakes and the littoral zone (shallow coastal zone) shall be considered shallow water disposal.

The waste material should be impounded within a confining structure which may be either natural or man-made, and designed to minimize the impounding area.

In the majority of instances where shallow water disposal in a lake is practiced, a dam or barrier should be constructed to confine the waste material. The structure of the dam must conform with sound engineering requirements. The dam should be constructed from materials which are not chemically reactive or exhibit any acid generating potential. Special consideration must also be given to:

- variable water levels on both sides of the dam;
- freeboard height on both sides of the dam;
- influence of wave action;

- permeability of the structure and the desirability of operating the tailings impoundment at a lower water level than the adjacent lake water to minimize seepage from the the tailings side;
- hydrology of the drainage basin discharging into the lake; and,
- dispersion of fines from the materials used for dam construction.

5.2.2 Lake disposal of tailings or waste rocks

In certain instances the essentially uncontrolled discharge of tailings into a lake may be proposed. While such an approach is not recommended it may have potential or merit for isolated lakes. Under these circumstances it may be necessary to reroute surface drainage and any through waters. In the event that a lake is used, the point of discharge from the lake should be considered the point of control for assessing compliance with regulatory requirements.

5.2.3 Deep water disposal of tailings

Where underwater disposal of tailings is used it is preferable in almost all circumstances to practice deep water disposal. Deep waters are considered those areas where the biological activity of the aquatic resource is at its minimum. In the marine environment this is generally below the 200 foot level and in inland waters this is generally considered to be below the 50 foot level. The greatest level of biological activity is associated with photosynthesis and below the levels referred to there is little light penetration and, consequently, little biological activity.

If deep water disposal of tailings is to be successfully practiced it is important to take into account the density and temperature of the water in the discharge zone. For example, in one case where marine disposal of tailings is being employed the tailings are partially dewatered in a thickener and the under-flow tails are repulped with sea water drawn from the 200 foot level, the depth at which the tailings are discharged. As a result adverse influences associated with the upwelling of low density current are minimized. In addition, the natural flocculating ability of the saline water assists in settlement of the solids.

Similar principles are considered in evaluating the disposal of tailings into lakes since natural stratification results in density differentials in the lake. Tailings should be dewatered, repulped with water drawn from the discharge zone and where required, treated with flocculants to promote the rapid and efficient settling of the fine fractions.

There are many unknowns related to the disposal of tailings and waste rock in both shallow and deep waters. These include the influence of a water cover on the acid generating potential of the waste materials and the influence of fine sediments on the aquatic life, especially the shellfish. The overriding concern is the problem of accessibility and controllability. Clearly, once materials are disposed of in either shallow or deep waters, they are inaccessible and any water pollution problems that develop cannot be controlled. With our present state of knowledge, regulatory agencies are adopting a prudent and conservative approach with respect to minimizing the disposal of tailings and waste rock in both shallow and deep waters.

5.3 Tailings Impoundment Operation

Many factors are of importance and must be considered in the successful operation of a tailings pond, including: size and depth of the settling pool; location and level of the pool; entry and decant locations and arrangements; wind action; short circuiting; freeboard height; and, exclusion of excess surface water from the pond, particularly uncontaminated water.

5.3.1 Pool size - *how big can be in settling*

The first requirement for efficient operation of a tailings impoundment is to maintain a pool of sufficient size to ensure adequate settling time and clarification of the decant or recycle water prior to its discharge from the pool. Although settling rates of various types and grain sizes of solids can be determined both theoretically and experimentally, these are of little use in themselves for the sizing of tailings ponds. The size of the pool required for satisfactory clarification of solids is determined by a number of considerations, including: fitness of particles; tendency to slime (clay type minerals); pH and temperature

of the water; wave action; depth of water; and, inlet and outlet arrangement and locations. In many instances where the tailings contain a substantial amount of slimes a considerable length of time is necessary to achieve satisfactory sedimentation.

Guides to insure clarification which have been proposed as a result of observation in existing ponds are:

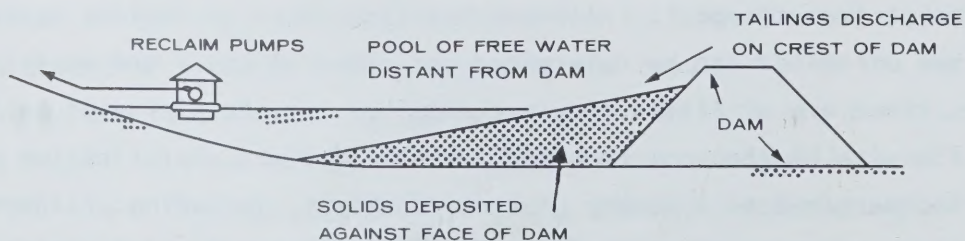
- 1) The pool should provide 5 days retention time and be designed to eliminate short circuiting.
- 2) The area of the pool should be sized to provide 10-25 acres for each 1,000 tons of tailings solids discharged per day. An average of 15 acres per 1,000 tons per day is usually considered adequate, unless some unusual conditions are present (13).

The settling characteristics of tailings can be improved, of course, by pH adjustment or the addition of flocculating reagents.

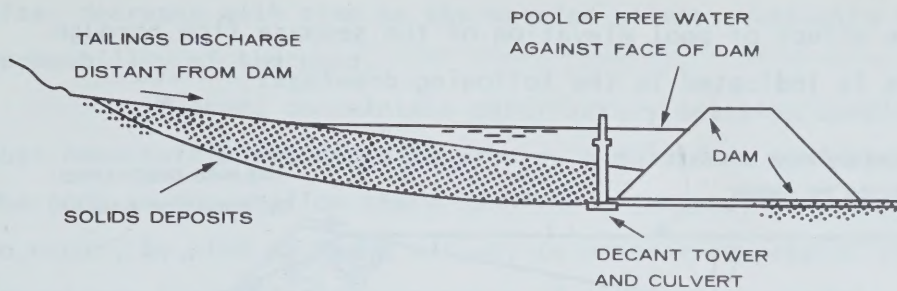
Where reagent stabilization is essential in the operation of a tailings pond this will dictate pool size, since retention time for adequate treatment is considered to be in the order of 30 days in most cases.

5.3.2 Location of Pool

The location of the pool of free water in the tailings impoundment is established, essentially, by the methods of discharging tailings into the pond. The following figures illustrate the effects of alternative tailings discharge systems. The first system involves the discharging from spigots or cyclones located on the crest of the tailings dam, and the second, from single point or multiple point feeding opposite the dam.



(a) DISCHARGE ON DAM



(b) DISCHARGE DISTANT FROM DAM

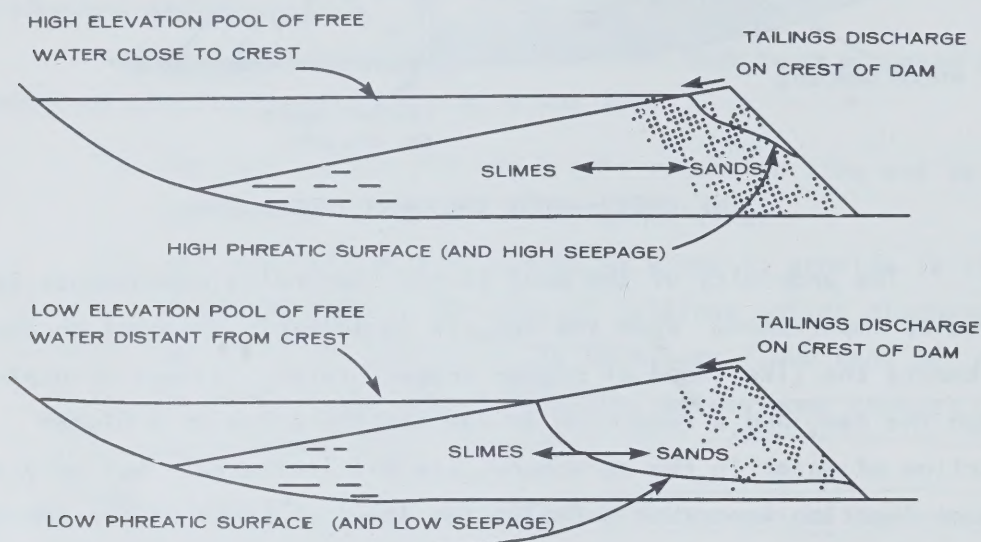
The proximity of the pool to the impounding embankments is of particular importance. When the pool is immediately adjacent to the embankments the likelihood of higher seepage rates, failure by piping through the dam, and a reduction in dam stability due to a higher proportion of water in the structure, are all increased. Not only is the pool location important but also the level of water at the embankment, since increased hydrostatic head will do nothing but further aggravate the problems mentioned above. The pool should never be allowed to overtop an unprotected embankment since failure can occur rapidly. Provision for a spillway of stable material is only prudent design under these circumstances. It is wise, therefore, that wherever possible the water in the tailings pool not be allowed to abut directly against the embankments, and where this is not possible, that the water level be kept to a minimum.

As evident in illustration (a) if the tailings are discharged by spigotting at several points along the dam, a layer or beach of coarse tailings will serve to separate the free water from the dam. This blanket of solids will reduce seepage flows through the dam and its foundations. In contrast, as shown in illustration (b), the discharge of tailings at points distant from the dam will result in the pool of water lying against the upstream face of the dam, leading to increased seepage rates and unfavourable influences on the stability of the dam.

5.3.3 Pool level

As previously mentioned, the level of the water in the pool can exert considerable effect on the seepage rate through the tailings dam and stability of the dam itself.

The effect of pool elevation of the seepage flow through tailings dams is indicated in the following drawings:



When the tailings are discharged from spigots located along the crest of the dam, the coarsest solid particles, as already mentioned, settle out closest to the dam and the finest, least permeable "slimes" settle farther out. At low water levels, the pool will be located some distance from the dam and over relatively impervious slimes. Consequently, the seepage rate and phreatic surface will tend to be low.

At higher water elevations the pool will be located closer to the crest of the dam and over the more pervious sands and, therefore, seepage and the phreatic surface will be higher with increased dangers of dam failure. Toe drains can be used to lower a high phreatic surface and thus prevent piping failures.

From the foregoing considerations of tailings dam seepage and stability, it is evident that the pool of free water should be kept as far back from the dam, and as low as possible. This, of course, must be considered in the light of providing adequate area for clarification of the pond effluent and reduced oxidation of sulphidic components in the tailings which could lead to further contamination in the tailings pond.

The tendency for seepage through the bottom of the tailings pond also increases with greater pool depth, although this seepage will

often decrease with time as the settled slimes eventually reduce the permeability of the pond.

However, to maintain satisfactory settling conditions the pool must have sufficient depth to provide a quiescent sedimentation zone. If the pool is too shallow there is often a tendency for sufficient turbulence to occur, by wind and wave action, to prevent successful clarification. In extreme situations resuspension of previously settled solids may occur. This is of particular importance near the decant outlets since insufficient opportunity would be available for resedimentation of the solids prior to discharge. Therefore, it is of prime importance to maintain sufficient water depth in the area of the decant outlets, especially since the velocity of the discharging water is increased as it funnels into the outlet. A minimum depth of five feet at the decant point is recommended.

5.3.4 Tailings discharge locations and arrangements

In order to provide the maximum opportunity for settling, and to reduce the chances for short-circuiting, the decant facilities should be located as far from the tailings entry points as possible.

Tailings are usually discharged through either a series of spigots located around the periphery of the embankment, or directly from the end of a pipeline, or by a combination of both methods. Cyclone feeding is used where it is necessary to separate a coarse fraction to be used for dam construction.

Spigot feeding of tailings offers numerous advantages including:

- 1) Tailings are distributed over a considerable area, which thereby reduces the number of pipeline relocations necessary.
- 2) Coarser tailings are deposited along the embankments and are available for further embankment construction.
- 3) Tailings enter the pond at a number of points and consequently the possibility of short-circuiting in the pond is reduced.
- 4) Material is available for separating the pool from the dam itself.
- 5) Better control of tailings distribution is attained.
- 6) Seepage rates may be reduced.

Cyclone feeding has the advantage of a greater degree of control over the size distribution of sands used for dam building. Due to the large volumes of fill used in the centre-line and downstream methods, this greater control over permeability usually must be exercised. In most cases this control is to ensure adequate drainage takes place, which in turn implies a well graded and relatively coarse construction material. The cyclones are usually mounted on the dam crest where considerable flexibility can be achieved in the location of the coarse and fine discharges. This method is generally not applicable to finely ground tailings since insufficient coarse material is available for dam construction. In one extreme case triple cycloning produced acceptable sands at a rate of only 15% of the total tailings.

Another method of limited application at the present time in Canada is the central discharge method. In this case, the tailings are thickened to the range of 30-40% solids and pumped to a central discharge point in the tailings impoundment area. The advantage is that greater control over the slope of the waste pile is possible. This is particularly important for fine, slimy materials whose natural cohesion is very low. It also keeps the tails away from the dam itself which may permit the use of a dam much lower in height than the final height of the waste pile itself. Provision must be made, however, to ensure the waste pile is stable (i.e. a lower angle of repose relative to the critical angle) or to provide enough residual freeboard at the dam to retain the material dislodged in the event of a shift in the waste pile. It is likely that this method will find increased application in the future as ore grades and grind sizes decrease.

5.3.5 Water reclaim systems

The following section is extracted from "Tentative Design Guide for Mine Waste Embankments in Canada" (13).

"1) Types

One of three basic systems is generally used for reclaiming or discharging clarified water from a tailings pond. A common system in the past has used "decant" culverts through the embankment. In this system, water near the surface of the pond flows through openings

spaced over the height of a tower, or riser pipe, constructed at some location in the pond. Each of these openings, in its turn, is plugged as the tailings in the pond reaches its level. Collected water flows down the tower and through a culvert constructed under the embankment. Downstream of the embankment the water can be allowed to drain away (if permitted by pollution control requirements) or it can be collected and pumped back to the ore processing plant.

A second water reclaim system now commonly used is to mount pumps on a floating barge or in a pumphouse capable of being moved up a ramp. These pumps draw water at shallow depths and discharge it into pipelines for return to the mill.

A third system uses siphon pipes installed over the crest of the embankment to draw water from the pond and discharge it to the downstream toe of the embankment.

"2) Comparison of Reclaim Systems

In relation to barge-pump reclaim systems, the decant system can have the following advantages:

- mechanical or electrical failures do not interrupt the discharge of water from the pond;
- if installed to cover the whole range of pond surface levels (from empty to full), the only recurring operation required to keep them in service is to successively plug the water intake openings as the tailings rise; and,
- if they have sufficient flow capacity, they can serve as permanent drains to handle runoff and keep the pond empty after the tailings operation has been abandoned.

They have the following disadvantages:

- where pond water is reclaimed from a point downstream of the embankment, rather than from the pond, the average pumping head to the mill is greater (assuming the pond is below the elevation of the plant);
- high decant towers are particularly susceptible to damage by movements of the tailings solids by which they are surrounded; destructive movements can be caused by slumping of the

tailings, particularly when they are deposited to substantially higher elevations along the edges of the pond than at points near the decant tower;

- the actual pressures to which decant culverts are subjected by overlying tailings are uncertain; in design, the conservative assumption, therefore, should be adopted that they are subjected to the full hydrostatic pressure of the saturated tailings above them;
- foundation settlements are very likely to crack and/or open joints in decant culverts, often leading to serious piping of tailings solids into and through the culvert pipes; and,
- even if large enough to permit a man's entrance, collapsed sections of culverts located beneath an embankment are very difficult to repair, and piping occurring into them is practically impossible to control.

The principal disadvantage of barge-pump reclaim systems is the necessity to move the barge (or pumphouse) periodically as the pond water surface rises. In order to keep these moves to a minimum the tailings disposal-reclaim system should be considered as a whole. Tailings sloping down from the disposal points should force the free water towards the barge-pump location. Where possible, the barge-pump should be located adjacent to a relatively steep portion of the pond perimeter so that long pontoon-supported flexible pipes and access ways can be avoided. Preferably, the barge should be moved away from the main embankment as the pond surface rises and expands in area. If practicable, this will permit the pond surface to be kept smaller, and farther from the embankment, for the same depth of water at the barge.

If used, decant culverts should be conservatively designed. Suitable design methods are described in the references. In general, they should be avoided where foundation settlements are expected. Barge-pump reclaim systems are better, in principle, from the point of view of embankment safety.

Siphon reclaim systems have been used on some tailings embankments. The main advantage of the siphon is its ability to pass full-capacity discharges with narrow limits of water surface rise in the pond.

Constructed over the crest of the embankment, it is subject to cracking by settlement of the embankment. Other disadvantages are:

- its inability to pass ice and debris;
- the possibility of water freezing in the inlet legs and air vents before the pond rises sufficiently to prime the siphon;
- the occurrence of surges and stoppages as a result of erratic make-and-break action of the siphon; and,
- the siphon pipes are subject to cavitation and low absolute pressures; for this reason, they are usually limited to a total drop of 20 feet maximum on earth dams."

5.3.6 Freeboard

The amount of freeboard between the pool level and the crest of the dam will depend on the location of the pool. Where it is against the dam a value of 2-3 feet is recommended. Where it is remote from the dam a freeboard of at least 1-2 feet is allowable provided the pond is at least 100 feet from the dam.

5.3.7 Seepage and stability criteria

The long term storage of tailings is a prime function of the tailings empoundment area. It is thus desirable to limit the amount of contaminated seepage released from the embankment and to ensure that structural stability is maintained. The two criteria are not always compatible. Control of seepage can be achieved by:

- constructing an impervious embankment;
- diverting surface drainage water around the embankment;
- covering the surface with an impervious blanket and grading the surface to encourage rapid surface runoff; and,
- revegetating the embankment to increase the rate of transpiration.

Ensuring a structurally stable embankment usually implies keeping a low phreatic surface in the embankment. This, in turn, requires a permeable embankment. There is no general agreement at this time as to the relative merits of impervious vs. permeable embankments. The use of surface drainage control and revegetation seems to reduce seepage from previous

dams to low levels although there are insufficient data available to state definitely if this will always be the case. An impervious embankment is structurally stable only as long as the impermeable membrane or core remains intact. Once breached, the embankment can liquefy if it is not designed to drain freely. At that point in time it would become a permeable embankment. The best technology suggests only that each embankment be designed on a case by case basis including all relevant stability criteria, both chemical and structural.

5.3.8 Diversion of surface drainage

Reducing the water flow into an embankment is essential for any type of construction used. This may require the construction of diversion ditches around the embankment. These should be designed on the basis of at least a one in 10 year storm and preferably longer if there is a chance that water overtopping the ditches can erode the embankment. Failsafe design should be used so that large shock water loadings can never compromise the structural stability of the waste embankment.

5.4 Lagoons

A treatment lagoon is very similar to a tailings pond. The main difference is that no solids tailings are deposited or stored in a lagoon. The other functions are the same as items 2-9 in Section 5.1. It can readily be appreciated that it would be difficult to fulfill all these functions in a single treatment lagoon and still maintain ideal conditions. For example, the accumulation of metal precipitates will ultimately reduce the volume of the lagoon available for sedimentation.

The advantages and disadvantages of lagoons are essentially the same as for tailings ponds. Sludge may accumulate in a lagoon, causing a reduction in its permeability and also posing a difficult problem when it must be removed for ultimate disposal. The iron ore industry uses lagoons (soak away pits) to dispose of excess process water. The action is essentially that of a sand filter and it removes residual solids to produce a clean effluent. Experience has shown that it is more economical in most designs to construct many lagoons of relatively small capacity and high settling efficiency than to construct one large, but inefficient, lagoon.

5.4.1 Acid neutralization

The use of lagoons for the treatment of acid waters was first employed in those locations where it was not convenient to direct the acid water to a tailings impoundment for treatment. Originally this was restricted to acid mine waters and surface runoff. More recently, acid waters seeping from tailings impoundments have also been directed to treatment lagoons.

The basic process considerations in the design of a lagoon are:

(a) Quantity of contaminated water - Mine water flows can be measured accurately through direct measurement or on the basis of pump characteristics. Fluctuations will occur on a seasonal basis for both underground and open pit operations, peaking during spring runoff. The interception of an aquifer will also affect the flow.

Seepage flows may be measured or estimated on the basis of the porosity of the dam. The influence of static head in the pond on flows should be predictable in a well engineered structure.

The volume of surface drainage will depend primarily on the location of the lagoon in the water shed. This is considered in Section 5.1.

The actual flow may be calculated from hydrological records and through the application of the following appropriate hydrologic soil complex members:

<u>Land Use</u>	<u>Hydrologic Soil Complex Number</u>
Bare Rocks	100
Lakes & Swamps	100
Residential	85
Silty Sand	90
Heavy Industrial	95
Native Grassland	71

The hydraulic design of the lagoon must be based on maximum expected flows. Generally a one in ten year flood is used. The quality characteristics of the surface runoff may indicate that peak

flood flows are uncontaminated and would not benefit from neutralization treatment. They may be diverted directly to a watercourse.

Alternatively a series of upstream dams and ponds may be used to contain flash floods and, with controlled pumping, provide flow equalization to the treatment lagoon.

General design approaches include a 2:1 length:width ratio and a minimum theoretical hydraulic retention period of three days. This is in excess of volume provided for sludge accumulation.

(b) Quality considerations - The atmosphere generally contains sufficient moisture for continued oxidation of exposed iron sulphide minerals. Thus, the first flush of flood water following a drought will often contain the highest concentration of contaminants. As the flood continues the supply of oxidation reaction products decreases and the contaminant level will drop. The ability to take advantage of this through diversion (see above) is dependant on local circumstances and must be individually assessed.

The quality of contaminated surface runoff may range over the values reported in Table 2. In non-sulphide ores (e.g. iron ore) the main contaminants will be excess suspended solids, nitrates and nitrites (from explosives) plus small amounts of oil and grease.

(c) Neutralization - For maximum process control and reagent effectiveness neutralization should be conducted and be complete before the reaction products enter the lagoon.

The basic process control parameter for neutralization is pH and it is essential that the pH be maintained at the level necessary for metal hydroxide precipitation. The pH and dissolved metal concentration of the waters will vary and the rate of reagent addition must respond accordingly. If the neutralizing reagents are blended with the acid water at the point of addition to the lagoon then accurate pH control of the lagoon contents is not possible. As an example, if the pH of the feed water is in the range of 2 to 3 a decrease in hydrogen ion concentration by a factor 10^7 may be necessary to precipitate zinc, nickel or ferrous iron. This cannot be achieved satisfactorily with a single pH sensing and reagent control system based on the feed water. Although a complementary

sensing system on the lagoon effluent may also be used to adjust reagent feed rates, the residence period in the lagoon would reduce the effective control response. A second factor to be taken into account is the reaction rate of the reagents. Even lime requires a specific contact time to fully exert its neutralization potential. Adding lime at the point of discharge to the lagoon will result in the deposition of unreacted reagent which will be lost. This results in unnecessarily high reagent costs.

Because of these factors a neutralization reactor is advocated. This may take the form of a completely mixed reaction vessel specifically constructed for the purpose or could be the pipes, launders and channels provided to bring the wastewaters to the lagoon. A reaction period in the order of ten minutes should be provided for lime. Due to the variable reactivity of different sources of lime and other neutralizing reagents, it is recommended that the actual reaction requirements be established on a case by case basis. Where warranted by raw water quality variations, a two stage pH control system is advocated. Reagent additions should be made on the basis of a two stage pH sensing and control system, with pH being measured in both the raw water and the neutralization reactor.

The reaction pH necessary for metal precipitation should be established on the basis of the metal species present (see Section 4.1.1) and should be augmented by laboratory and/or pilot scale studies. In those instances where lime is used, the reduction in pH on standing due to recarbonation must be taken into account. This may be done by providing a reaction pH one unit higher than that indicated by theoretical considerations and short term bench scale studies.

5.4.2 Sedimentation

Process requirements for sedimentation can readily be established in conventional tube settling tests. A major problem exists in attempting to relate the settling characteristics in a tube to those prevailing in a lagoon. The actual flow regime in a lagoon is extremely variable and subject to the influence of wind, temperature, inlet and outlet location, design, ice conditions and short circuiting effects, etc. These factors make it impossible to advocate settling design considerations on a purely rational basis. However, a number of empirical observations can be made. These are illustrated in Figure 11.

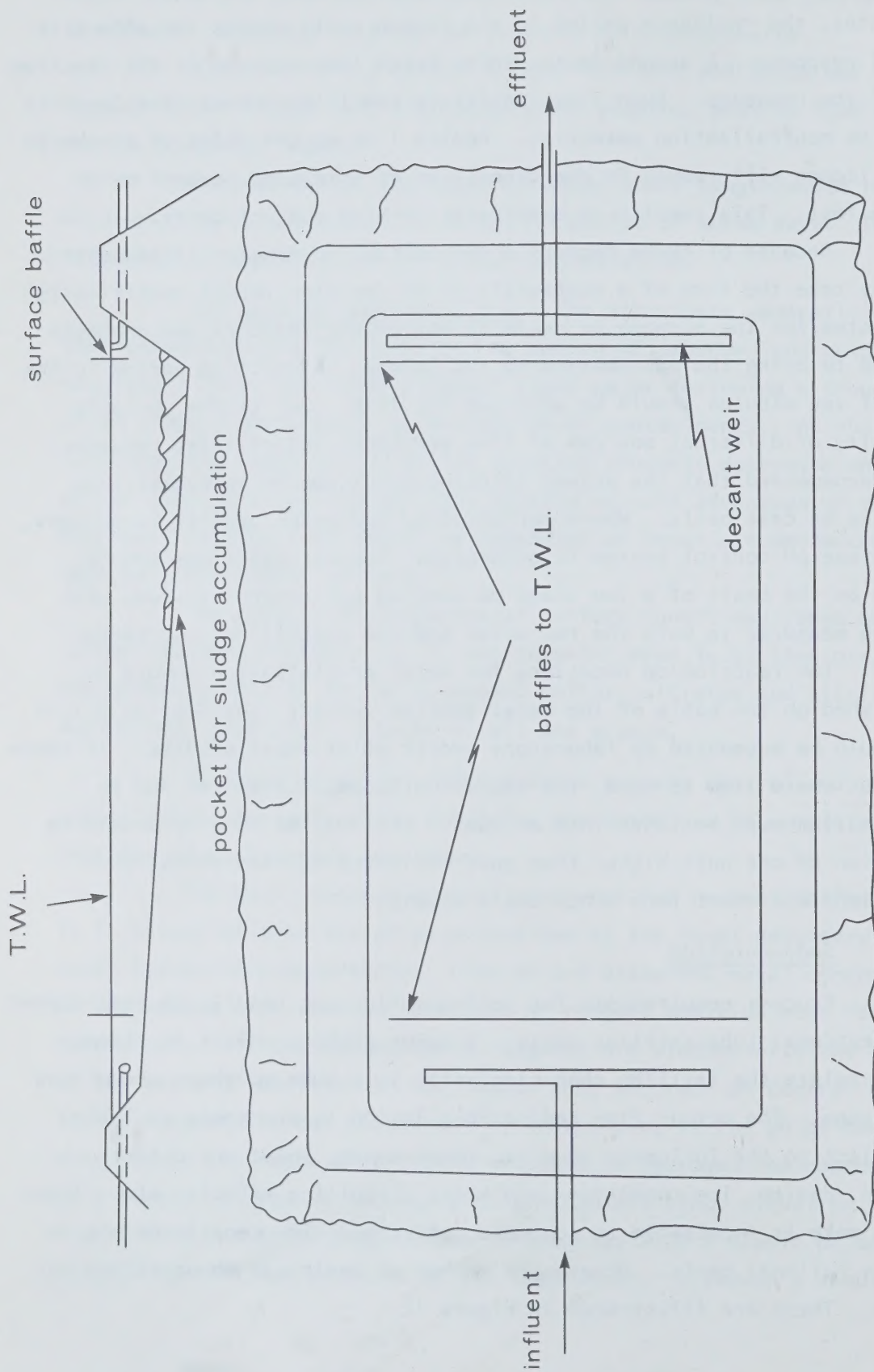


Figure 11. Schematic Representation of Treatment Lagoon.

- 1) The inlet should be as far as possible from the outlet. The kinetic energy of the feed water should be dissipated and the water distributed uniformly as far as possible over the complete cross section of the lagoon. A T-shaped inlet configuration with different inlet orifice areas can provide this. Flow velocity and current density are frequently neglected in theoretical studies of sedimentation lagoons. Entering velocities are difficult to dissipate, particularly in large lagoons, and may even build up to create tornado-like "storms" among settling particles.
- 2) Baffling can be used to insure that the complete volume of the lagoon is used for sedimentation and surface skimming is minimized. In certain cases, such as the soak-away lagoons, sectioning the entire lagoon into a number of small ones can prove to be beneficial.
- 3) The overflow should be collected over a wide cross section of the lagoon rather than at a single point.
- 4) Specific provisions should be made for sludge accumulation so that resuspension of previously settled material is minimized.

An overriding consideration in the use of a lagoon is that metal precipitates should be readily settleable. As previously discussed in Section 4.5, precipitates formed from waters low in metals may be dispersed and exhibit poor settling characteristics. These may be enhanced through the techniques advocated above.

5.4.3 Sludge accumulation and disposal

The principle of making special provisions for the accumulation of sludge has been introduced above. Although this is a very basic and obvious design consideration it is frequently overlooked, with the result that an otherwise well engineered lagoon fails and has an effective life of only a few months. This constitutes a very expensive lack of foresight.

The volume of sludge produced is dependant on the amount of suspended solids, the acidity of the water, the dissolved metal concentration and the neutralizing reagent used. In treating concentrated wastewater,

up to 30% by volume of the feed can report as sludge. This is at the maximum density of 2-3% by weight solids found in the simple neutralization process. Obviously, this can constitute a very major problem. Solutions may be provided by either allowing sufficient volume for total sludge accumulation or by sludge removal and alternate disposal.

Allowance for perpetual sludge retention is possible where:

- 1) the water is low in metals and suspended solids and thus would have a small sludge yield;
- 2) the source will release contaminants for only a finite period, e.g., underground mine water;
- 3) a very large lagoon is available (in certain instances lakes have been sacrificed for this purpose); and,
- 4) high permeability is not required (as in soak-away pits).

It must be cautioned here that sludge accumulation in a lagoon possesses a basic inherent disadvantage. If pH control in the neutralization process is lost then the potential for redissolving accumulated sludges exists.

The physical problems associated with sludge removal are immense. The gelatinous, thixotropic nature of the material precludes the use of pumps. Draglines have been used but they represent a barely workable, expensive, emergency solution.

One technique which has indicated promise is the practice of underlaying the sludge with a network of collector pipes. At the time of sludge removal, compressed air is sparged into the sediments to resuspend them and the fluidized material is then withdrawn. Although this removes the sludge from the lagoon the resultant fluid still represents a major disposal problem. The stability of the sludge when exposed to rainwater (see Section 4.5.2) limits surface disposal of these materials. Naturally, when a lagoon is being desludged alternative means of treatment must be employed.

5.4.4 Ferrous/ferric oxidation

The stoichiometric requirement for the oxidation of ferrous iron to ferric iron is 0.14 lb O_2 /lb of ferrous iron. The associated oxygen transfer rates cannot be achieved by diffusion from the atmosphere

into a quiescent body of water unless the lagoon is either extremely shallow or the initial ferrous concentration is very low and under alkaline conditions. Oxygen transfer requirements can be met by forced aeration. The induced turbulence would usually indicate separate facilities for sedimentation.

5.4.5 Radium removal

The process requirements for radium removal have been described in Section 4.8. The recommended operational practices are similar to those discussed in Section 5.4.1.

5.4.6 Thiosalt stabilization

5.4.6.1 Biological oxidation. In spite of the distribution and significance of the thiosalt problem in Canada there are only two mining operations employing full scale thiosalt stabilization processes. They both employ a biological oxidation method although different approaches are taken.

(a) Case #1

A lead/zinc milling operation uses liquid SO_2 in the flotation circuit. A mill effluent concentration of 1000 mg/l thiosulphate is typical, with approximately equal proportions attributed to the oxidation of sulphide minerals in an alkaline grinding circuit and to the use of SO_2 during flotation. This apportionment of thiosalts is subject to confirmation.

A lagoon is provided, downstream from the tailings pond, with a thirty day theoretical volumetric residence period. After bacterial oxidation the decant flows to another lagoon of similar design. The feed to the second lagoon is dosed with lime to provide an ultimate design pH of 9.5.

During warm weather a thiosulphate reduction of 80% is consistently obtained. No data are available on conversion to polythionates or sulphates. During the winter months the system operates less satisfactorily with thiosulphate reductions of less than 20 per cent. This reduction in efficiency is largely attributable to the effects of temperature on the oxidation process (see Section 4.2.2). The restricted availability of oxygen and short circuiting because of ice cover may be additional factors.

Consideration of temperature in process kinetics indicates that biological lagoons will not be capable of providing a consistent effluent quality under Canadian winter conditions.

(b) Case #2

The influence of temperature has been compensated for at the other Canadian mining operation with a thiosulphate treatment process. In this instance, a ponding system with a residence period of one year is provided. Influent thiosulphate concentrations are attributable only to the milling of sulphide ores and are reported in the range of 200 - 300 mg/l. Essentially complete thiosulphate stabilization is reported with effluents characterized by COD values of less than 1 mg/l in comparison with feed strengths of 50 mg/l.

Neutralization is effected by limestone, wet ground to minus 325 mesh, to a pH of 7.2. Metal values, principally copper and iron, are reduced to satisfactory levels indicating oxidation of ferrous to ferric iron.

5.4.6.2 Chemical oxidation. The limits placed on the application of biological oxidation processes for the treatment of thiosalts have led to increased interest in chemical oxidation. Basic process considerations have been discussed in Section 4.2. Test work is currently underway to evaluate ozone, and also possibly chlorine, on a pilot scale. Tentative cost data for both concentrated, 1000 mg/l, and dilute, 100 mg/l, thiosulphate waste streams are given in Tables 12 and 13.

5.4.7 Effluent polishing

Both tailings impoundments and lagoons are generally not capable of producing a consistent effluent quality. Factors adversely affecting effluent quality include wind, precipitation, ice formation and spring runoff. These have been considered above and in Section 4. Practical modifications to the design and operation of tailings ponds can rarely counteract these adverse influences and, as a consequence, an additional effluent polishing system is advocated. This would ensure that the final effluent from the property would be of a consistent quality equivalent to the performance of the primary system under ideal conditions.

TABLE 12. COMPARISON OF CHEMICAL TREATMENTS FOR THIOSALTS (9)
 (Based on treating a 2 mgd stream of 1000 mg/l S_2O_3)

Chemical	Quantity Required per lb S_2O_3 (lb)	Estimated Cost/lb (\$)	Daily Cost (\$)	Estimated Relative Capital Cost of Facilities	Neutralization Required
<u>Oxidants</u>					
Ozone	1.06	0.16 0.125	3800 2500	High High	Yes Yes
Chlorine	3.16	0.03	1900	Low	Yes
Calcium Hypochlorite	3.22	0.40	8000	Low	Questionable
<u>Buffers</u>					
Sodium Bicarbonate	1.5	0.0465	1400	Low	---
Soda Ash	0.95	0.075	1425	Low	---
<u>Neutralizing Agents</u>					
Limestone	0.89	0.0001	18	Low	---
Lime	0.50	0.001	100	Low	---

TABLE 13. COST OF TREATING WATER CONTAINING THIOSALTS (14)

(Based on treating 1 mgd of water containing
100 mg/l of S_2O_3)

Compound*	Reagent Cost (¢/lb)	Weight Used Daily at 100% Efficiency (lb)	Daily Cost (\$) Efficiency of Use	
			100%	70%
Chlorine	5	1,270	64	91
Ozone	17	172	29	42
Sodium Chlorate	7	1,520	106	152
Ozone **	17**	1,070	182	260
Hydrogen Peroxide (35%)	52	3,500	630	900

* All prices F.O.B. manufacturer except for ozone which is produced on site.

** Including amortization, cost of labour, maintenance and power.

A secondary treatment system would use as feed the effluent from a tailings pond or treatment lagoon. Basic design considerations would be as discussed in Section 5.3.

A secondary treatment system based on lagoons may be considered as a means of consistently achieving the best effluent quality attained by the primary system when that is operating under ideal conditions. The working group considers this to be a basic objective for mine wastewater treatment systems.

5.5 Ponding of Seepage Waters

The ponding and natural oxidation of acid seepage waters has been reported to have considerable treatment potential. Results have indicated complete ferrous iron oxidation and reductions of up to 50% in dissolved sulphate and trace metals. It has been suggested that the formation and sedimentation of a jarosite type material may be the dominant factor in the purification process although this has to be confirmed.

The requirements for the process are:

- (a) a retention time in the order of 6 months;
- (b) the pond should be as shallow as possible (to enhance oxygen transfer from the atmosphere);
- (c) the pond waste should not be allowed to come into contact with material containing iron sulphide minerals; and,
- (d) wastes being decanted from the pond should be taken from the pond surface (preferably a sluice type overflow).

The surface layer of the pond is the zone of maximum oxidation and should thus contain the most stabilized material.

This technique is most suited to the treatment of concentrated flows of low volume, such as seepage waters from abandoned areas. Although local consideration of topography etc., will limit its applicability, the principle is considered to have general application.

The pond effluent may require further treatment for acid neutralization and metal removal before it is released to the environment. However, the natural ponding process represents a means of achieving a substantial savings in reagent costs for neutralization.

5.6 Full Scale Mechanical Treatment Plants

The use of mechanical equipment in mine wastewater treatment processes is in its infancy in Canada. At the present time the only full scale mine wastewater treatment plant in Canada is being operated by Chester Mines in New Brunswick. Treatment incorporates neutralization, clarification, sludge recycling and effluent polishing in a 24-hour lagoon. The 200 lpgm plant and its operation have been described by A. V. Bell (15). The basic plant flowsheet is shown in Figure 12. The mine drainage to be treated was anticipated to contain approximately 1600 mg/l acidity (as CaCO_3), 1500 mg/l sulphate, 300 mg/l iron, 100 mg/l copper and 100 mg/l zinc. The plant has been in operation since November, 1974 and although the strength of the plant influent has not reached the strength expected, the effluent discharged after treatment has been averaging 0.01 mg/l copper, 0.09 mg/l zinc, 0.09 mg/l lead and 0.2 mg/l iron.

The plant clarifier (25 ft diameter) is not housed and it is interesting to note that it operated satisfactorily during the winter even with its surface covered by a 12 inch layer of ice.

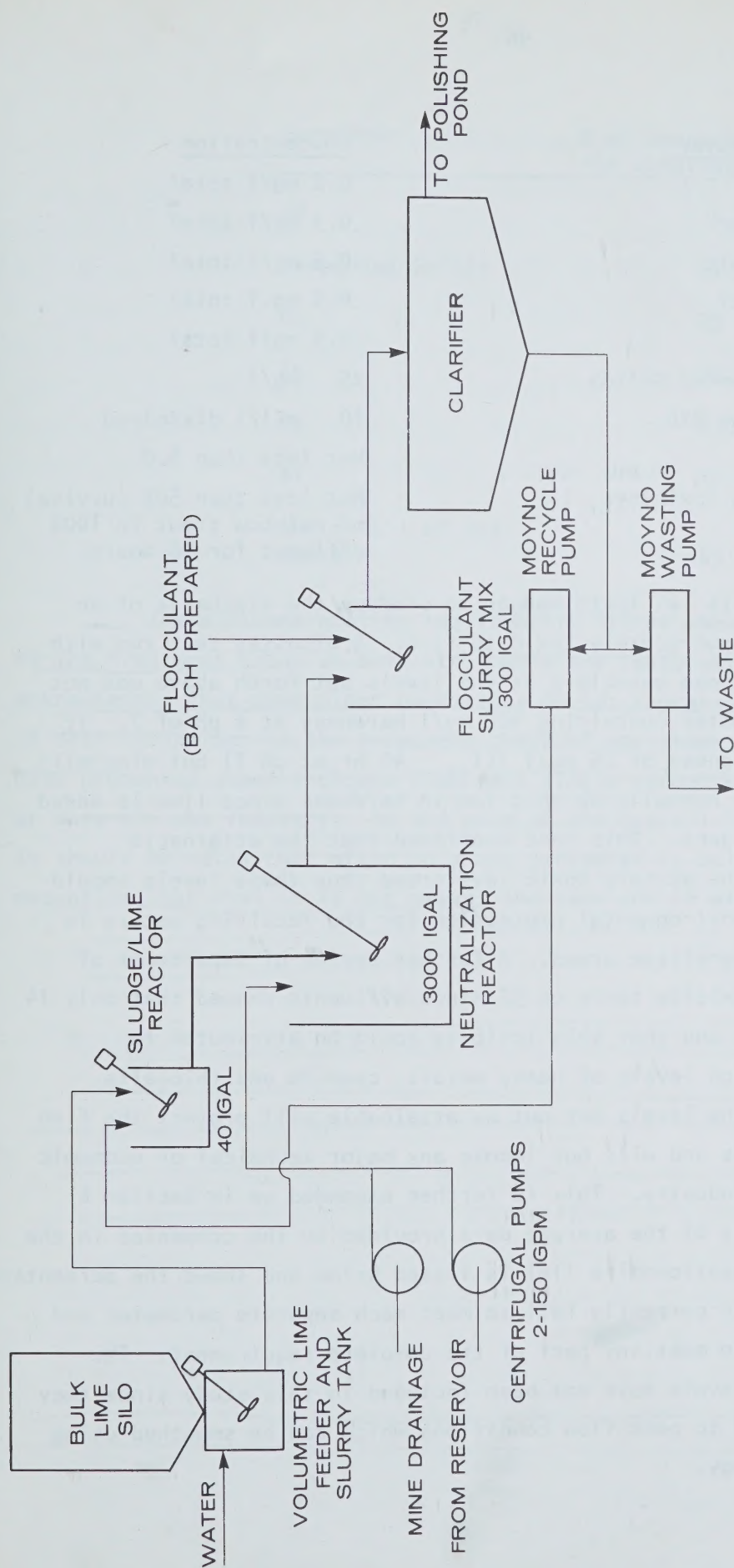
The capital cost of the plant was approximately \$110,000 in 1974. Operating costs, including provision for an operator for one shift/day, are approximately \$0.70/1000 Imp. gallons mine drainage treated.

At the present time, several larger treatment plants are in either the design or construction stage. The largest is a 6,000,000 lpgd facility scheduled for start-up by International Nickel Company at Sudbury early in 1976. This plant will include two clarifiers and employ sludge recycle.

5.7 Attainable Effluent Quality

The monthly average* levels of effluent quality attained through the application of best practicable technology should not exceed the following total concentrations:

* Arithmetic monthly average is based on the sampling and analysis schedule as set out in the Mining Effluent Guidelines and Regulations (in preparation). Radium 226 is determined on a dissolved metal basis using a 3-micron filter before the determination.



NOTES

DESIGN CAPACITY: 200 IGPM AT 1600 MG/L ACIDITY AS CaCO₃

- 1 ALL TANKS AND CLARIFIER OF MS CONSTRUCTION COATED WITH COAL TAR
- 2 ALL MIXERS PROPELLER TYPE
- 3 CLARIFIER - 25' DIA x 10' MS SIDEWALLS
- 4 ALL EQUIPMENT EXCEPT CLARIFIER SHELTERED IN PREFAB 24' x 48' BUILDING

Figure 12 Chester Mines Treatment Plant Basic Flowsheet

<u>Parameter</u>	<u>Concentration</u>
Lead	0.2 mg/l total
Copper	0.3 mg/l total
Arsenic	0.5 mg/l total
Nickel	0.5 mg/l total
Zinc	0.5 mg/l total
Suspended Solids	25 mg/l
Radium 226	10 pCi/l dissolved
pH	Not less than 6.0
Acute Toxicity	Not less than 50% survival of rainbow trout in 100% effluent for 96 hours.

The limits set forth appear to produce the discharge of an effluent which is non-acutely toxic to fish. A bioassay test run with water containing those materials at the levels set forth above was not acutely toxic in water containing 400 mg/l hardness at a pH of 7. It was toxic at a hardness of 25 mg/l (LT_{50} - 40 hr at pH 7) but mine-mill effluents will not normally be this low in hardness since lime is added as a treatment reagent. This test confirmed that the attainable levels are below the acutely toxic levels and thus these levels should provide adequate environmental protection for the receiving waters in all but the most sensitive areas. A further series of Department of the Environment toxicity tests on 57 mines effluents showed that only 14 were acutely toxic and that this toxicity could be attributed to the presence of high levels of heavy metals, cyanide and thiosalts. It is clear that the levels set out as attainable will protect the fish resources of Canada and will not impose any major technical or economic hardships on the industry. This is further expanded on in Section 6.

A summary of the average data provided by the companies in the Mine Wastewater Questionnaire (16) is listed below and shows the percentage of operations which currently fail to meet each separate parameter and also the failure to meet any part of the complete requirement. The maximum effluent levels have not been included in this study since they tend to be related to peak flow conditions which can be smoothed using available technology.

Parameter	% of operations not meeting the attainable levels
pH	5.1
Suspended Solids	21.5
Cu	29
Zn	22.7
Pb	13.6
Ni	38.4
Any or all of the above	65.4

The effluent quality requirements listed above are regarded by the Treatment Group as both attainable and reasonable. The industry acknowledges that some older operations do not represent the application of best technology to the treatment phase of operations. The percentage data presented above indicate that most single parameters are met by 70% or more the the industry. In the case of the overall failure percentage it should be noted that often only one parameter is below standard for any operation, but that it is not always the same one in all cases.

6. THE COST OF WASTE MANAGEMENT (17)

The information presented in this section is designed to provide a concise and stepwise approach to obtaining order-of-magnitude costs for the segregation, transportation and treatment of metal mining effluents.

The section is intended for use only as a "ball-park" cost estimating procedure. It is not intended to serve as a design manual and it is assumed that the user either possesses, or has access to, the necessary engineering expertise required to apply many of the functions presented.

Due to the variations in mine, mill and effluent characteristics encountered across Canada and the nature of the procedure, the accuracy of costs obtained by these methods should not be assumed to be greater than 30%. This does not detract from the usefulness of the procedure if the user bears in mind the over-all philosophy of an order of magnitude estimate and its inherent limitations.

It has been assumed that the following information is available prior to applying the estimating procedure:

- 1) the type of mine under consideration and its capacity;
- 2) basic rock and ore characteristics especially their acid generating potential;
- 3) contour plan of area; and,
- 4) some qualitative or preferably quantitative data on local stream quality.

Basic steps suggested as an overall procedural approach to assessing the costs of a particular drainage control and treatment scheme are:

- a) Obtain or assume the above mentioned data.
- b) Sketch out the broader aspects of the development and the alternative to be costed including:
 - (i) proposed locations of all mine head and open pits;
 - (ii) proposed locations of waste rock piles;
 - (iii) proposed locations of mill as well as material handling and storage facilities;
 - (iv) proposed location of tailings impoundments; and,
 - (v) alternate receiving stream for treated effluents.

- c) Indicate the drainage areas controlled or influenced and designate each as potentially contaminated or uncontaminated.
- d) Indicate the drainage segregation alternatives.
- c) Follow costing procedures presented in the following section for:
 - (i) surface drainage control;
 - (ii) pumping and piping;
 - (iii) tailings impoundments; and,
 - (iv) waste treatment.

6.1 Drainage Control

There are two important aspects commonly involved in the control of surface drainage at a mine site:

- segregation and collection of contaminated drainage; and,
- diversion of uncontaminated streams around working or contaminated areas.

These must be considered separately due to the different magnitude of drainage area normally associated with each and the assumptions that can, therefore, be made from the hydrologic standpoint.

This section provides the means of obtaining "ball-park" estimates for surface drainage diversion works. Pumping costs, which are often involved in the control of surface drainage, are considered separately in Section 6.2.

6.1.1 Contaminated drainage control

6.1.1.1 Ditches. Areas from which contaminated drainage must be collected seldom exceed one square mile. For areas of this magnitude, or less, it is reasonable to assume total runoff of the design storm (24 hr, 10 year recurrence) in 24 hours. The estimating procedure to obtain diversion ditch costs is as follows:

- (i) determine one in 24 hr storm of 10 yr recurrence interval (or other assumed design storm);
- (ii) enter contaminated drainage area on Figure 13 and obtain excavation cost/ft run as shown; and,

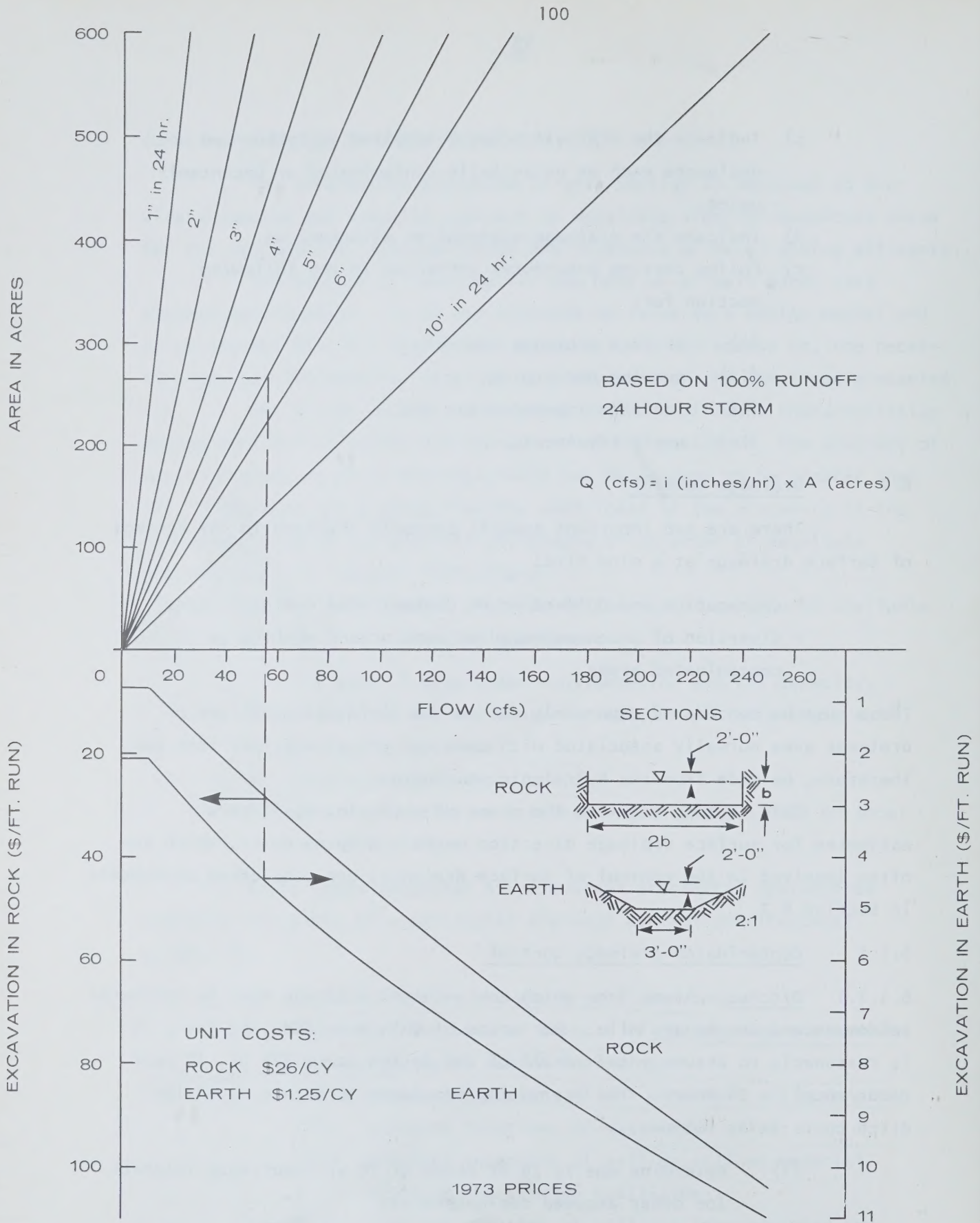


Figure 13. Surface Drainage Diversion Channel Costs

- (iii) compute total excavation cost using estimated length of ditch.

For special sections, e.g. combined earth and rock excavation, use the upper portion of Figure 13 to obtain flow and evaluate excavation cost based on a calculated cross section (assume velocity of 3 fps and earth excavation costs of \$26/cu yd).

6.1.1.2 Holding ponds. Control of contaminated surface drainage often involves the construction of holding ponds to collect and balance flow. It has been assumed that these will normally be constructed of homogeneous borrowed fill type dams not exceeding 30 ft in height. The cost estimating procedure is as follows:

- (i) determine storage volume required;
- (ii) from topographic maps choose location and estimate average dam height and length to provide volume required;
- (iii) use Figure 14 to determine dam construction costs;
- (iv) add 20% for foundation preparation, cut offs etc.;
- (v) refer to Section 6.2.2 for pumping cost estimates;
- (vi) allow \$50 per cfs for small concrete overflow spillways; and,
- (vii) use lower portion of Figure 13 to estimate for seepage collection ditches where applicable.

6.1.2 Uncontaminated drainage diversion

6.1.2.1 Small drainage areas. Segregation of clean drainage from areas of less than one square mile entails essentially the same procedure as those described for contaminated drainage. The costing procedure described in Section 6.1.1.1 should, therefore, be used.

6.1.2.2 Stream diversions. Where it is necessary to divert streams controlling drainage areas greater than one square mile the same assumptions cannot be made to determine the flow volumes involved. The design flow must be determined using regional streamflow data, or from rainfall data, taking into account the hydrologic characteristics of the region. Where this information is not known, the reader is referred to the applicable regional office of the Water Survey Branch of Environment Canada for representative streamflow data (cfs per sq mile) suitable for rough estimating procedures.

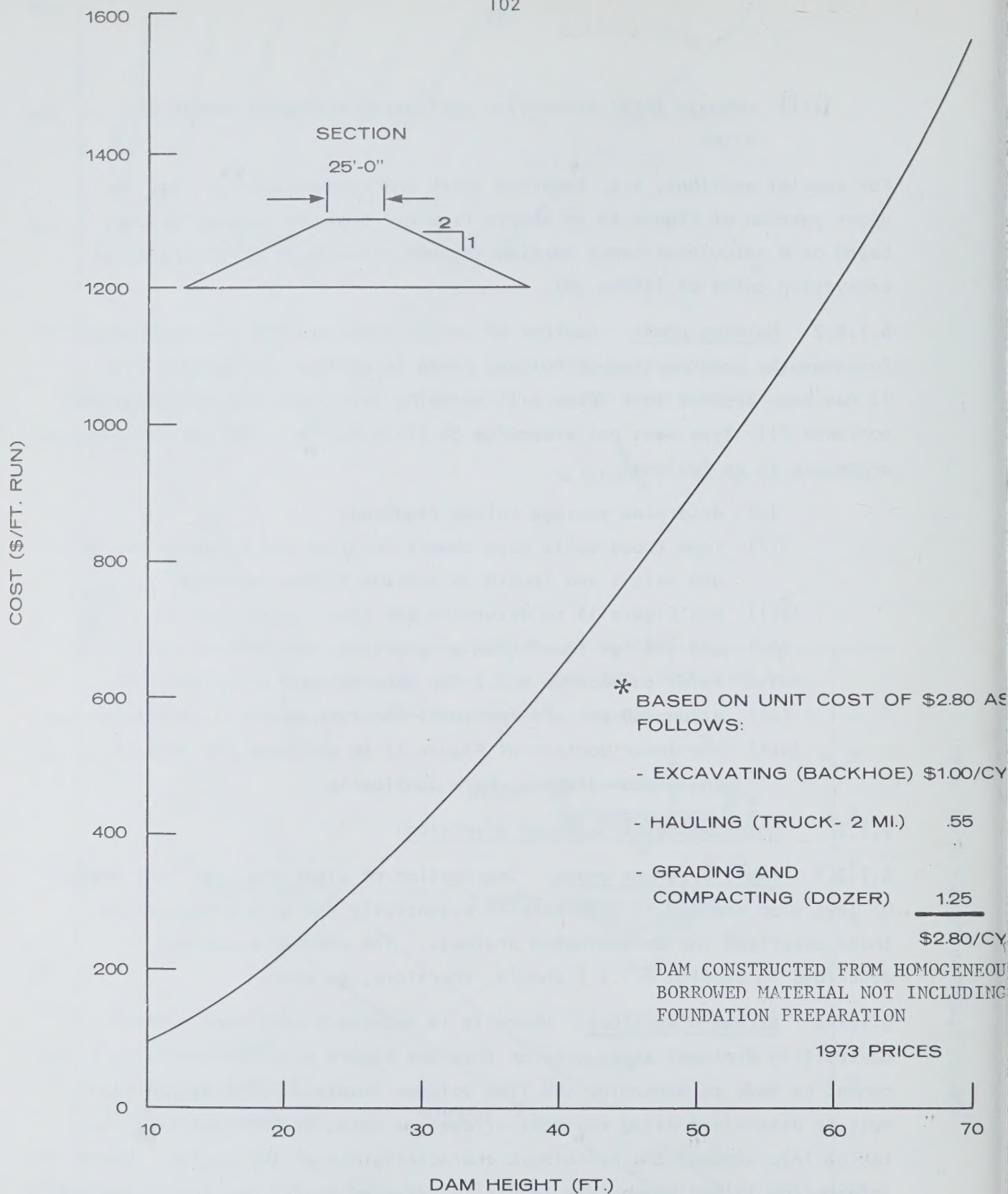


Figure 14. Small Dam Costs *

Estimating procedure is as follows:

- (i) determine the peak stream flow to be diverted;
- (ii) use Figure 15 to determine diversion channel costs;
- (iii) use Figure 14 to allow for diversion dams where necessary or calculate cost of concrete diversion structures on basis of \$200/cu yd in place;
- (iv) for minor access bridges over diversion allow \$20,000 cost installed:

24" dia.	\$10.00/lin. ft
36" dia.	\$20.00/lin. ft
48" dia.	\$27.00/lin. ft
60" dia.	\$40.00/lin. ft
72" dia.	\$58.00/lin. ft; and,

- (v) add cost of excavation and backfill required according to unit prices given in Figure 15.

6.2 Pumping and Piping

The data and procedures provided in this section enable "ball-park" estimates to be made of capital and operating costs for piping clean water, contaminated drainage and tailings slurries, together with the installation costs of various pipe support methods, insulation, etc.

The following information is required to apply the procedures in this section: flow rate, head, distance, specific gravity of liquid or slurry, and type of pipe material required.

6.2.1 Piping costs

Given the above mentioned information, the procedure for estimating the cost of piping is as follows:

- (i) determine pipe diameter using nomograph on Figure 17 and from this directly obtain cost per ft of pipe for chosen material;
- (ii) allow approximately \$1,000/acre for right-of-way clearing and grubbing where necessary;
- (iii) allow up to \$10/lin. ft for routine grade preparation for pipes laid on-grade;

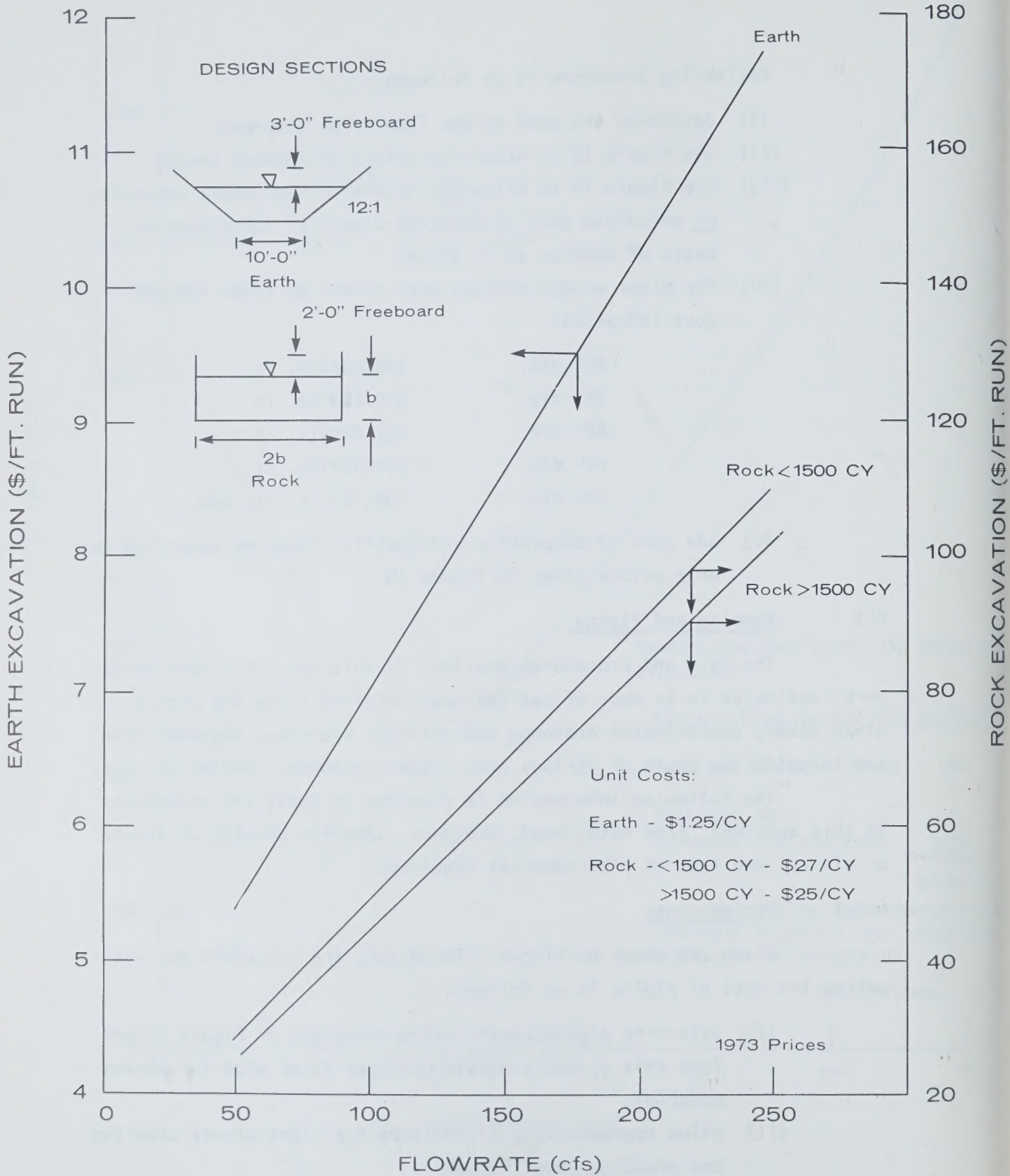


Figure 15 Diversion Costs - Small Streams

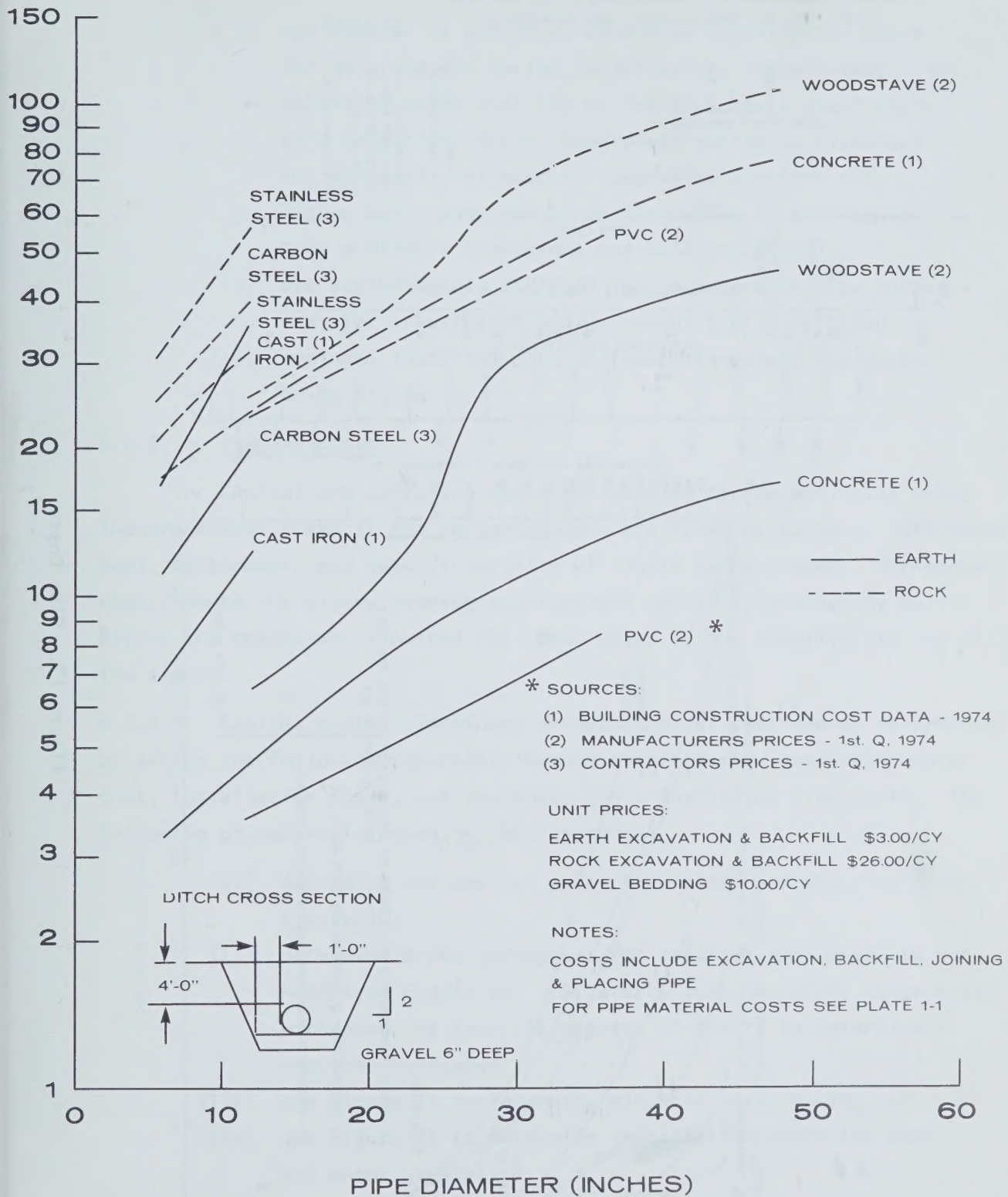


Figure 16. Below Ground Installation Costs - Pipe



Figure 17. Pipe Costs

- (iv) use Figures 16 and 18 to determine installation costs for on-grade and buried installation, respectively. (It should be noted that the buried pipe costs given represent good construction conditions making no allowance for difficulty of access, dewatering problems etc. Higher unit rates should be used where it is suspected that difficult conditions may be encountered);
- (v) use approximately \$50/foot run to determine pipe support cost for trestles of medium height (20-30 ft); and,
- (iv) evaluate installed cost of insulation where applicable using Figure 19.

6.2.2 Pumping costs

Capital and operating costs for pumping can be estimated using the procedures given if the following data are known or assumed: discharge head, horsepower, and specific gravity of liquid being pumped. Horsepower requirements for slurry pumping applications can be determined by multiplying the horsepower required for clear water by the specific gravity of the slurry.

6.2.2.1 Capital costs. Total approximate capital cost can be determined by adding the following separately determined factors: pump cost, motor cost, installation costs, and pumphouse and transmission line costs. The following procedure is based on this approach:

- (i) determine the cost of pump, base-plate and coupling from Figure 20;
- (ii) determine brake horsepower for required pump capacity and head from Figure 20. For slurry pumps multiply clear water horsepower by specific gravity of slurry to determine required horsepower;
- (iii) use Figure 21 to determine electric motor costs;
- (iv) use Figure 22 to determine installation costs for pump and motor assembly;
- (v) using approximate unit cost of \$50/ft² installed, calculate cost of transmission line (assuming 12 k.v. wood



Figure 18. On-Grade Pipe Installation Costs

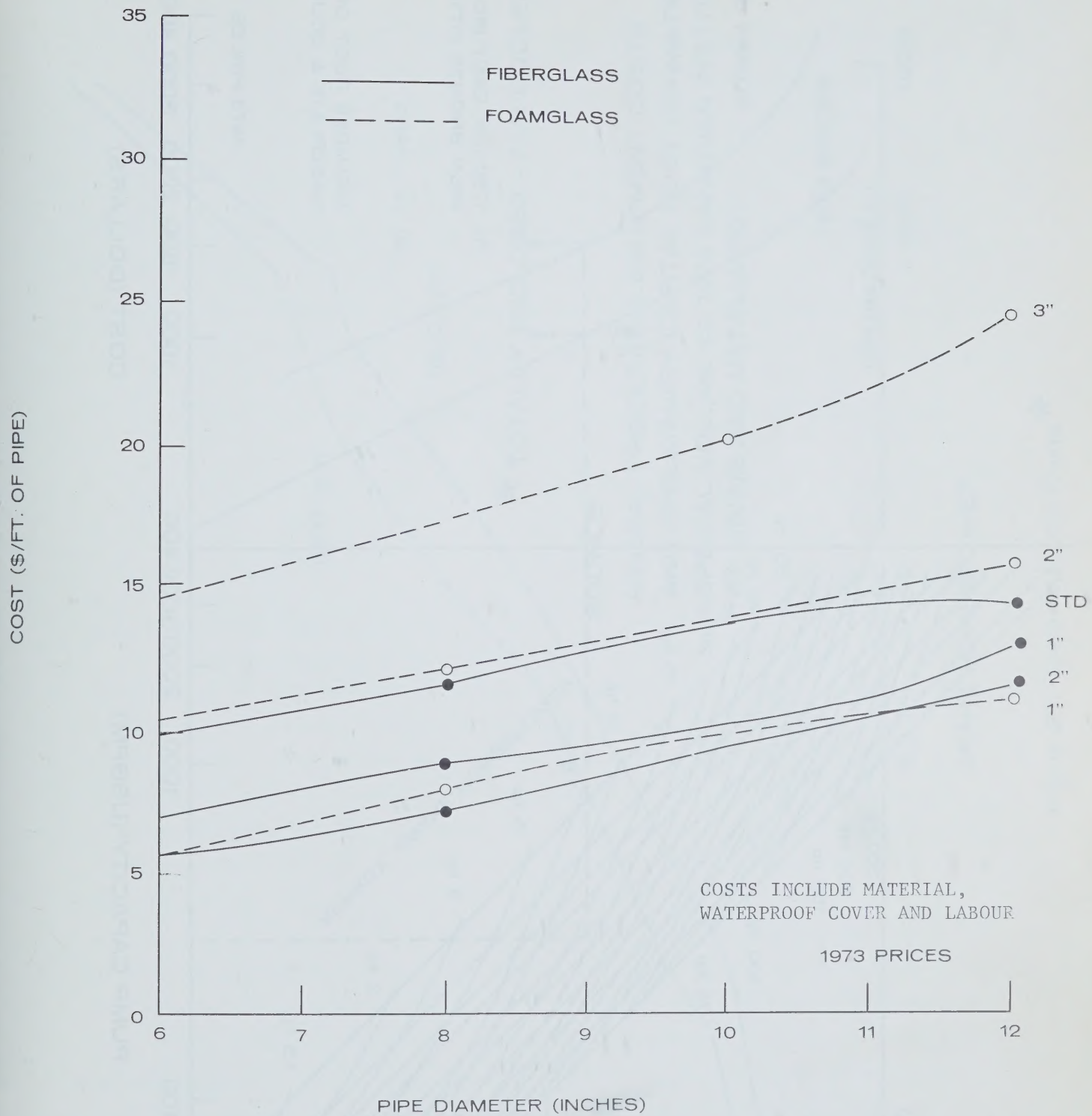


Figure 19. Cost of Pipe Insulation

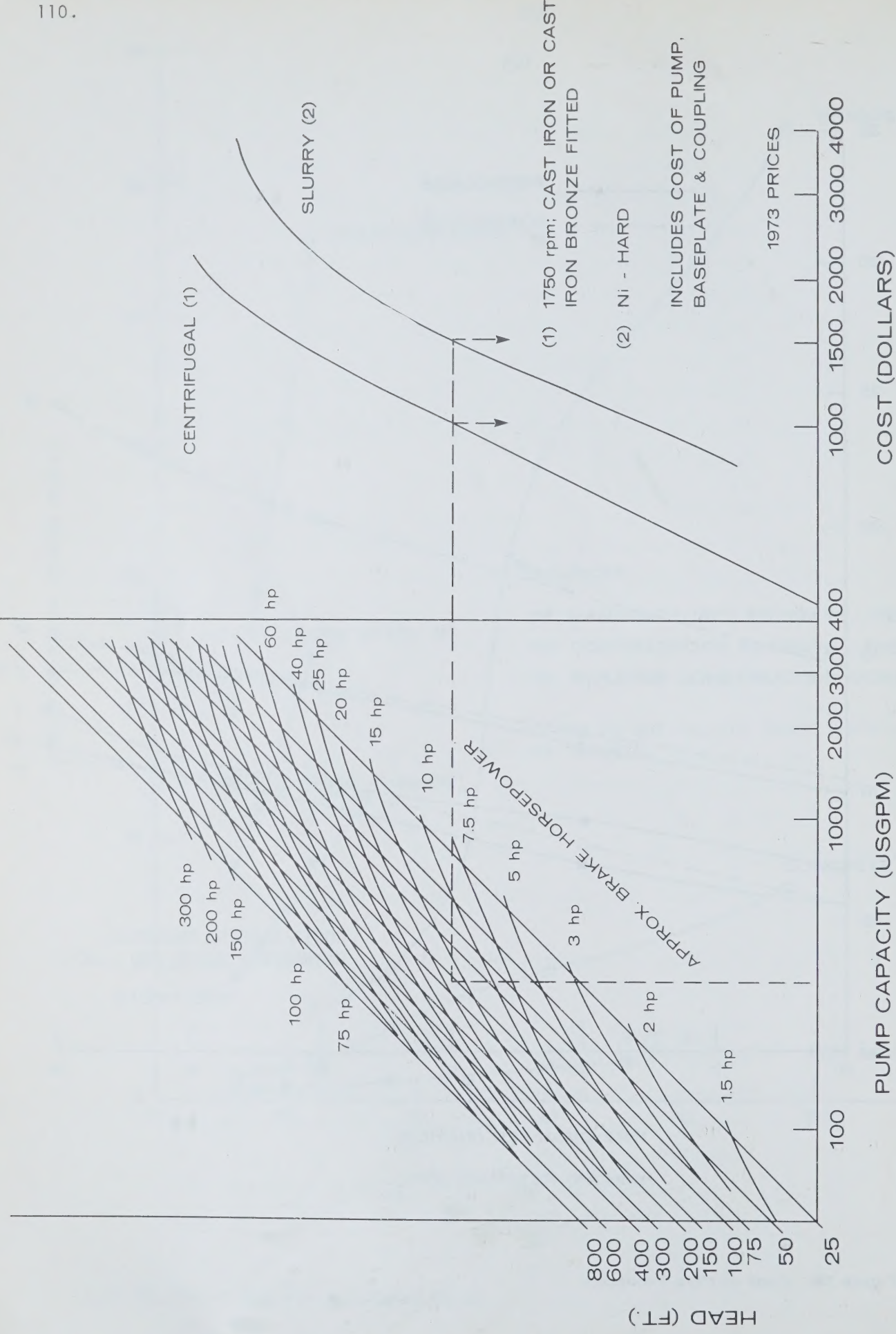


Figure 20. Pump Costs

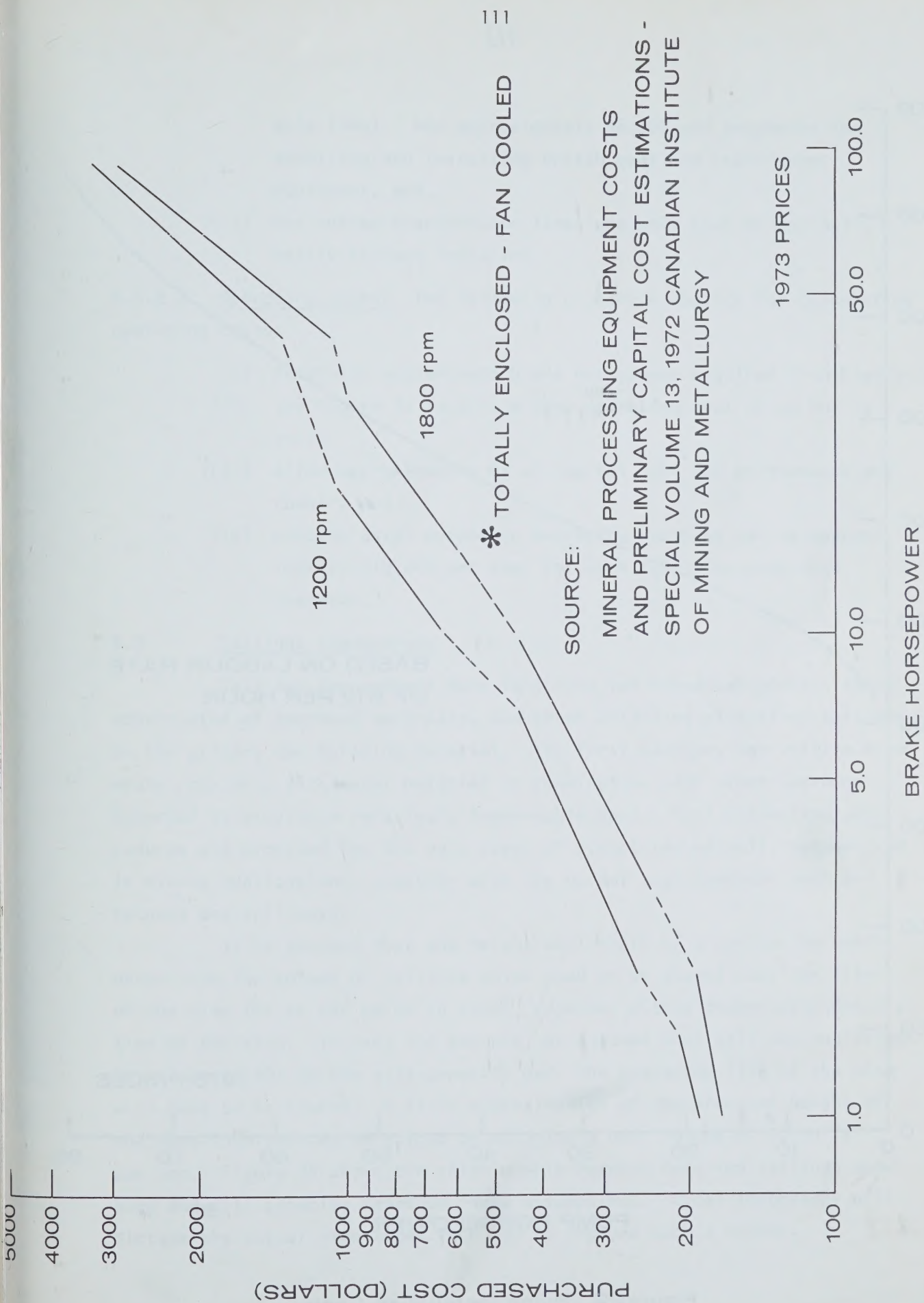


Figure 21. Electric Motor Costs *

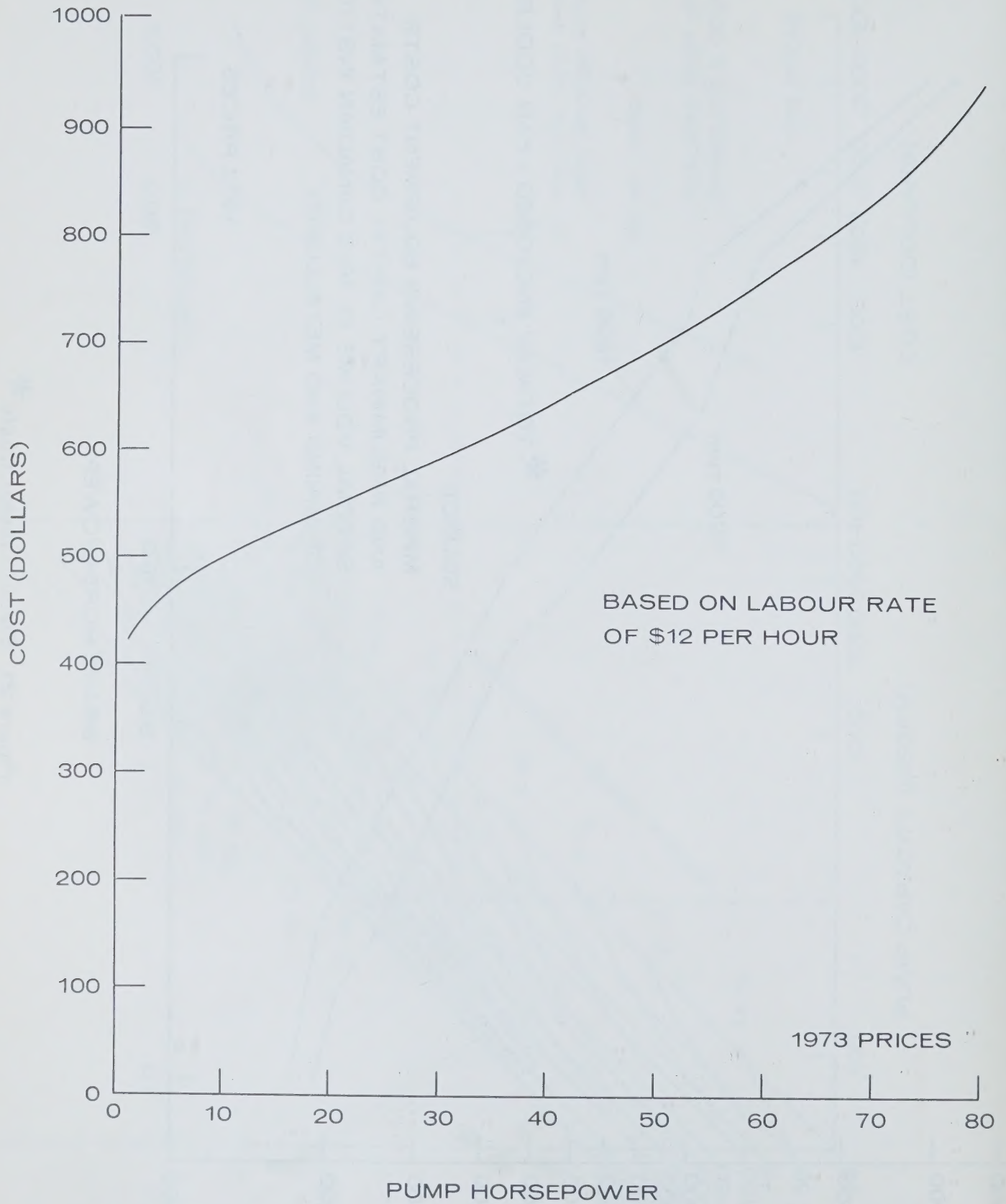


Figure 22. Pump Installation Costs

pole line). Add approximately \$8,000 per pumphouse for supplying and installing switch gear and transformer equipment; and,

- (vii) for buried transmission lines use unit cost of approximately \$4/foot installed.

6.2.2.2 Operating costs. The following procedure applies for calculating operating costs:

- (i) determine approximate brake horsepower required from Figure 20;
- (ii) use Figure 23 to obtain pump operating cost on an hourly basis;
- (iii) allow approximately 6% of capital cost for maintenance and repair; and,
- (iv) compute total pumphouse operating costs by adding approximately \$10,000 per year for each full-time pumphouse operator.

6.3 Tailings Impoundment

Tailings impoundment dams fall into two broad categories: those constructed of borrowed materials, and those utilizing classified tailings as the primary dam building material. The first category may utilize mine waste rock as a structural material in combination with other borrowed material to provide a relatively impermeable seal. Cost estimating procedures are provided for the main types of structures normally encountered in mining applications, together with the normal appurtenances such as decants and spillways.

It is assumed that the height and length of a dam can be evaluated from the volume of tailings which need to be stored over the life of the mine (or at any point in time), together with a topographic description of the area. It can, for example, be assumed that tailings equivalent in volume to 80% of the mill capacity over the projected life of the mine will have to be stored. A first approximation of the area and height of the dams involved can be gained by assuming a unit volume of 20 cu ft per ton. Figure 24 shows the relationship between required tailings pond area and mill capacity based on these assumptions. Local topography will dictate the actual structures required to impound such a volume.

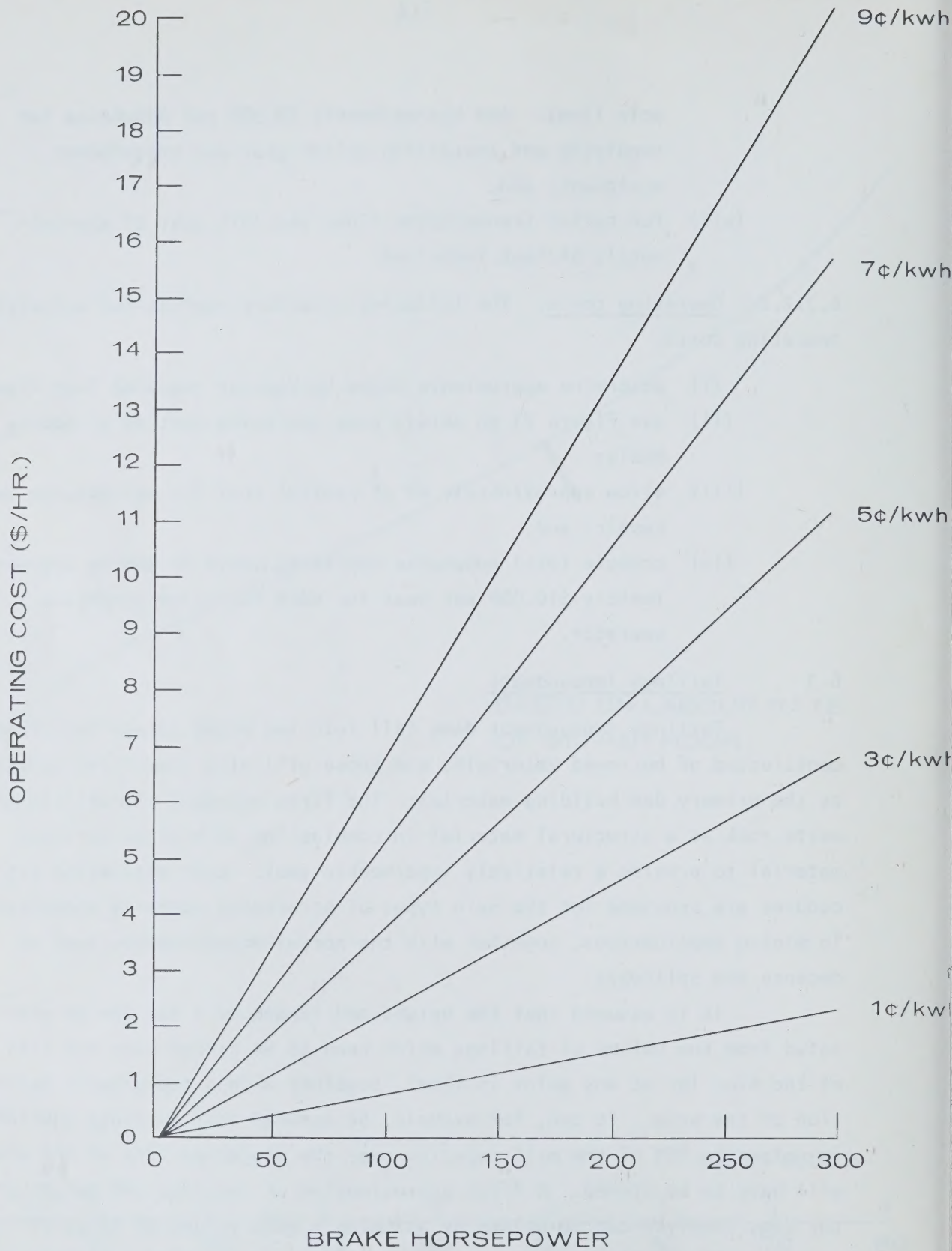


Figure 23. Pump Operating Costs

MILL CAPACITY (TPD)

TAILINGS POND AREA (ACRES)

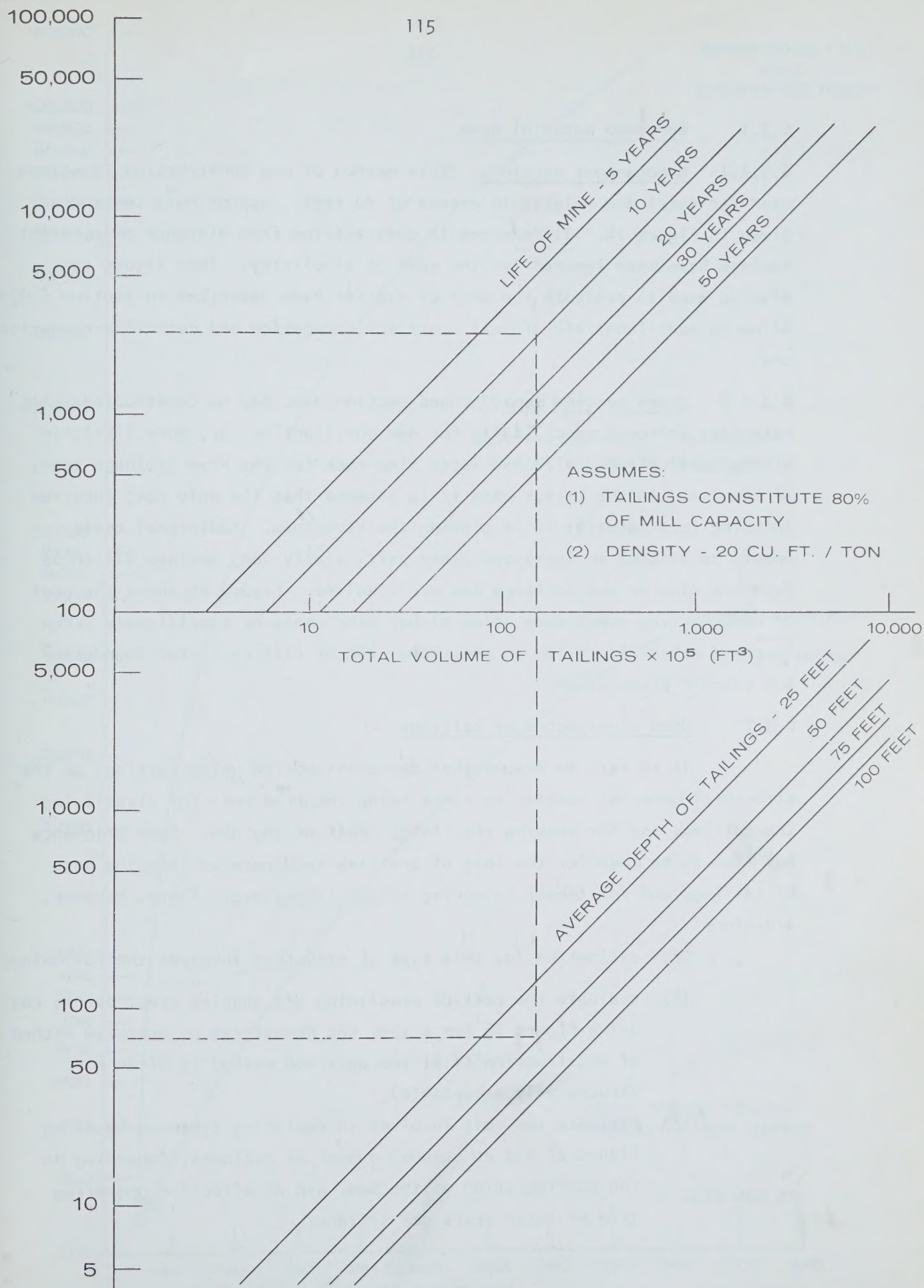


Figure 24. Mill Capacity VS Tailings Pond Area

6.3.1 Borrowed material dams

6.3.1.1 Homogeneous sections. This method of dam construction is seldom used for total dam heights in excess of 70 feet. Approximate costs are given in Figure 14. Differences in cost arising from distance of material haulage have been ignored for the sake of simplicity. This figure can also be used to evaluate the cost of starter dams described in Section 6.3.2. Allow an additional 20% of full costs for foundation and cut-off preparation, etc.

6.3.1.2 Zoned section dams. Zoned section dams may be constructed using materials borrowed specifically for dam construction, or, more likely in mining applications, will use waste mine rock for the free drainage zones of the dam. In the latter case it is assumed that the only cost incurred in using this material is in placing the structure. Additional costs should be allowed in instances where particularly long haulage distances from the mine to the tailings dam are involved. Figure 25 shows the cost of constructing zoned dams using either mine waste or specifically borrowed material. Allow an additional 20% of fill costs for foundation and cut-off preparation.

6.3.2 Dams constructed of tailings

It is fair to assume that dam construction using tailings as the structural material results in costs being incurred only for classifying the tailings and for shaping the rising crest of the dam. Some allowance has also to be made for the loss of trestles incorporated into the mass of tailings and for labour in moving spigot lines, etc. These, however, are minor.

Cost estimation for this type of structure involves the following:

- (i) evaluate the cost of draglining and shaping crest per ft run using Figure 26 for either the downstream or upstream method of construction (N.B. the upstream method is often not structurally acceptable);
- (ii) estimate the cost involved in replacing cyclones based on Figure 27 and an assumed number of cyclones, depending on the configuration of the dam, and an effective operating life of three years per cyclone;

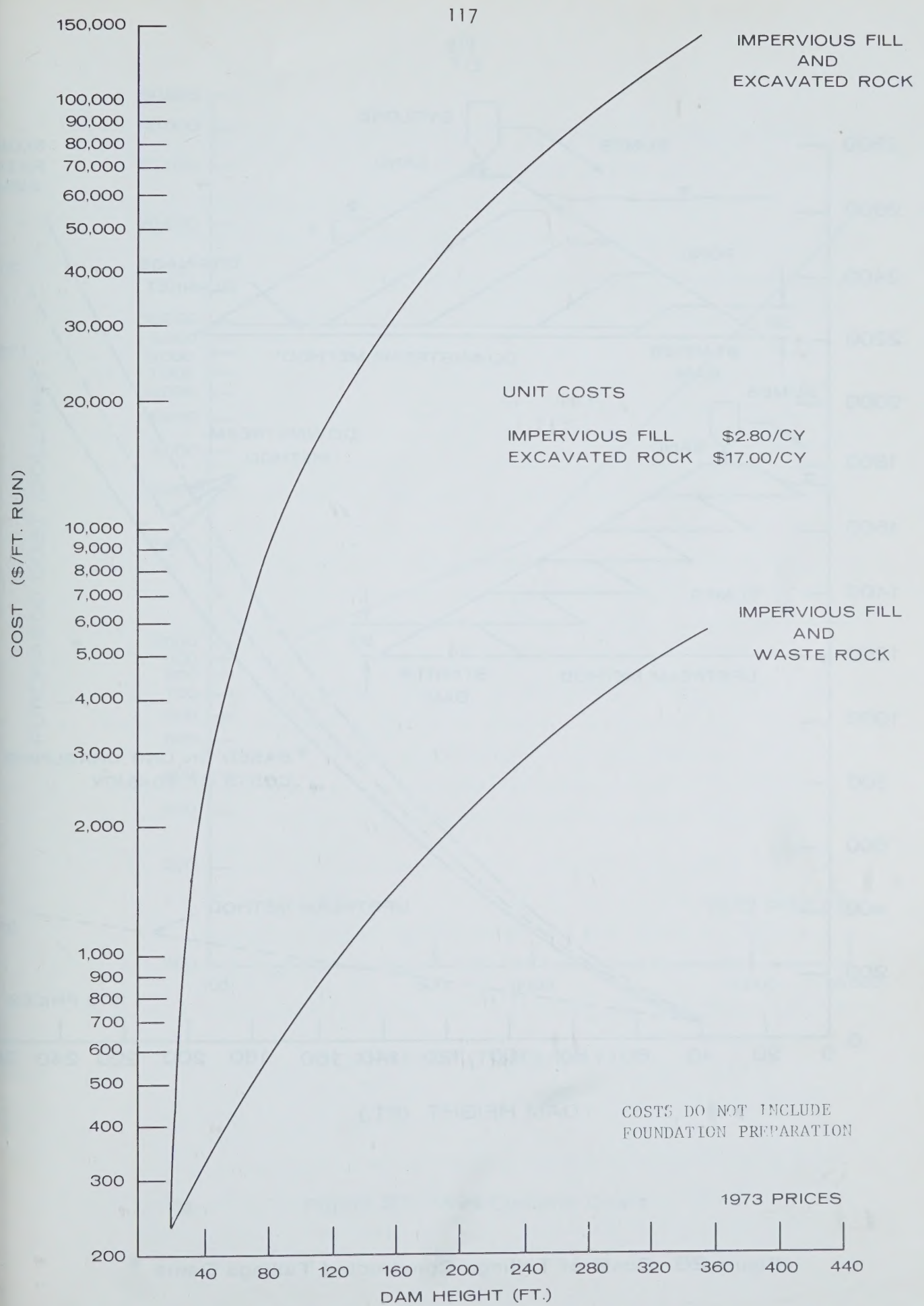


Figure 25. Cost of Zoned Dams

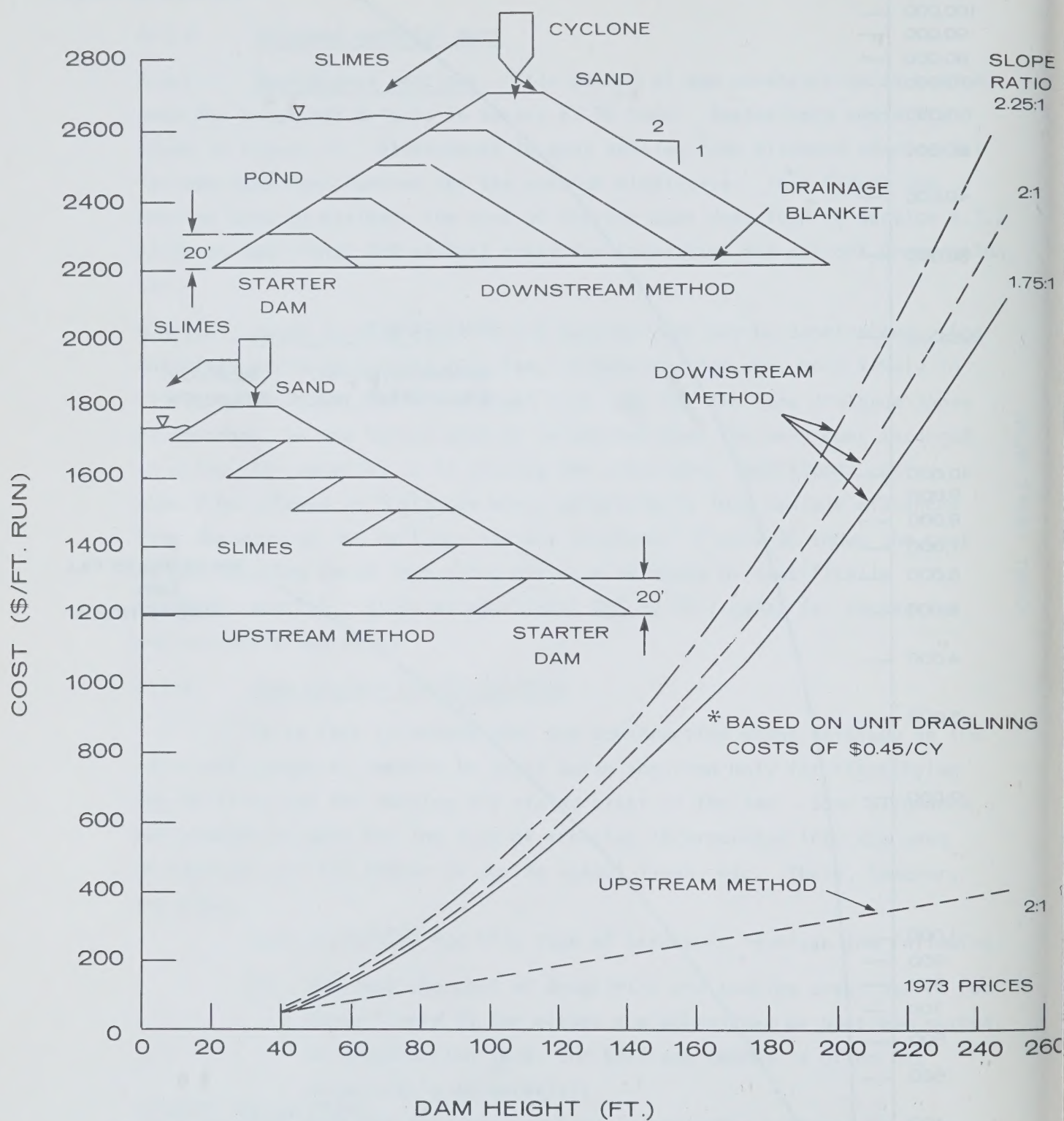


Figure 26. Costs of Tailings Constructed Tailings Dams *

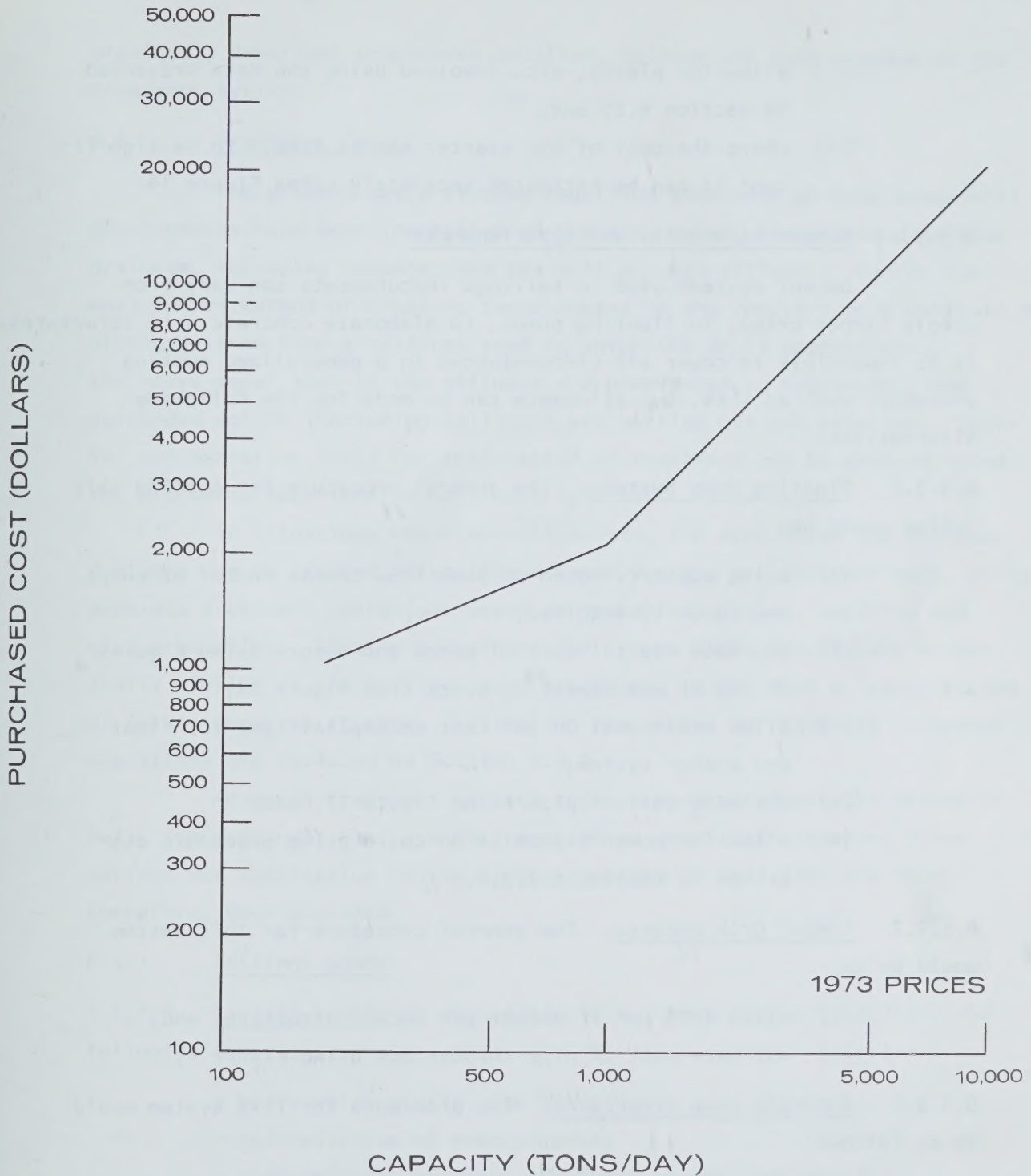


Figure 27. Wet Cyclone Costs

- (iii) allow for piping, etc. involved using the data presented in Section 6.2; and,
- (iv) where the cost of the starter dam is likely to be significant it can be estimated separately using Figure 14.

6.3.3 Decant structures and appurtenances

Decant systems used in tailings impoundments can vary from simple timber cribs, to floating pumps, to elaborate concrete drop structures. It is impossible to cover all circumstances in a generalized costing procedure such as this, but allowance can be made for the following alternatives:

6.3.3.1 Floating Pump Systems. The general procedure for costing this system would be:

- (i) using capacity based on peak flow choose number of pumps and capacity required;
- (ii) estimate capital cost of pumps and motors using Figures 20 and 21 and operating costs from Figure 23;
- (iii) allow additional 20 per cent on capital cost for float and anchor system;
- (iv) estimate cost of pipe using Figure 17; and,
- (v) allow for power transmission costs using procedure described in Section 6.2.2.1.

6.3.3.2 Timber Crib Decants. The general procedure for this system would be to:

- (i) allow \$100 per ft height per decant structure; and,
- (ii) estimate cost of pipe through dam using Figure 17.

6.3.3.3 Concrete Drop Structures. The procedure for this system would be as follows:

- (i) allow \$200 per cu yd for poured concrete; and,
- (ii) estimate cost of pipe through dam using Figure 17.

6.3.4 Seepage collection

It is usually important to collect seepage from the toe of a tailings impoundment and return it to the system of treatment. Use

previously described procedures to ditch, collect and pump seepage to the treatment system.

6.4 Treatment Costs

The primary waste streams requiring treatment at base metal/mill developments have been identified as the mine water, contaminated surface drainage, including seepage, and the mill process effluent. By far the most common method of treating these wastes in the industry as a whole is to discharge them into a tailings pond in which the pH is controlled, the heavy metal ions in the effluent are precipitated as hydroxides, and suspended solids (including tailings) are settled out and retained. Capital and operating costs for this method of treatment can be derived using the procedure described in Section 6.4.1.

In situations where no mill exists, the mine water and surface drainage can be treated using lagoon based systems or, alternatively, using separate treatment operations based on mechanical mixing, settling and sludge handling. Costing procedures for lagoon based systems are essentially similar to the tailings pond procedures except that a lesser volume of solids requires storage. Estimating procedures for separate treatment operations are included as Section 6.4.2.

Advanced treatment methods such as reverse osmosis, electrodialysis, ion exchange, etc. are not regarded as practical treatment alternatives for application in the mining industry at this time and have, therefore, been excluded.

6.4.1 Tailings ponds

6.4.1.1 Capital costs. A well designed tailings pond will perform the following functions:

- heavy metal precipitate formation;
- sedimentation of precipitates;
- sedimentation of other suspended solids (tailings);
- final retention of solid wastes (tailings and precipitate sludges);
- stabilization of oxidizable constituents e.g. ferrous to ferric, thiosalts to sulphates, organic reagent residuals, etc.;

- balancing influent quality and quantity fluctuations; and,
- storm water storage and balancing.

Consequently, few, if any, supporting unit processes are necessary except under special circumstances. Therefore, capital costs for tailings pond treatment systems involve only the construction costs of the dams and appurtenances already outlined in Sections 6.3.1 to 6.3.3. Approximately \$30,000 should be added to this estimate to cover costs of neutralization reagent storage and handling facilities for typical installations.

6.4.1.2 Operation costs. Operating costs for tailings pond treatment facilities may be estimated using the following procedure:

- (i) use Figure 28 to estimate neutralization reagent costs per 1000 U.S. gal/min treated. (Note that reagent costs are based on residual acidity after mixing all waste components);
- (ii) determine total pump and pumphouse operating costs by summing operating costs for each pumping installation as outlined in Section 6.2.2.2;
- (iii) allow \$10,000 per annum operating labour costs; and,
- (iv) approximately 6% of the total capital cost should be included for maintenance and repair.

6.4.2 Separate treatment operations

Mine waste treatment systems based on separate unit operations will normally involve the following:

- alkaline reagent slurrying and feeding;
- flocculation and sedimentation; and,
- sludge filtration and disposal.

6.4.2.1 Capital costs. It is beyond the scope of this report to present a detailed cost breakdown of each of the above-mentioned unit operations. The approximate capital cost of a lime neutralization/sedimentation plant can be derived using Figure 29 although the limitations of applying a general set of overall cost curves for specific applications must be borne in mind.

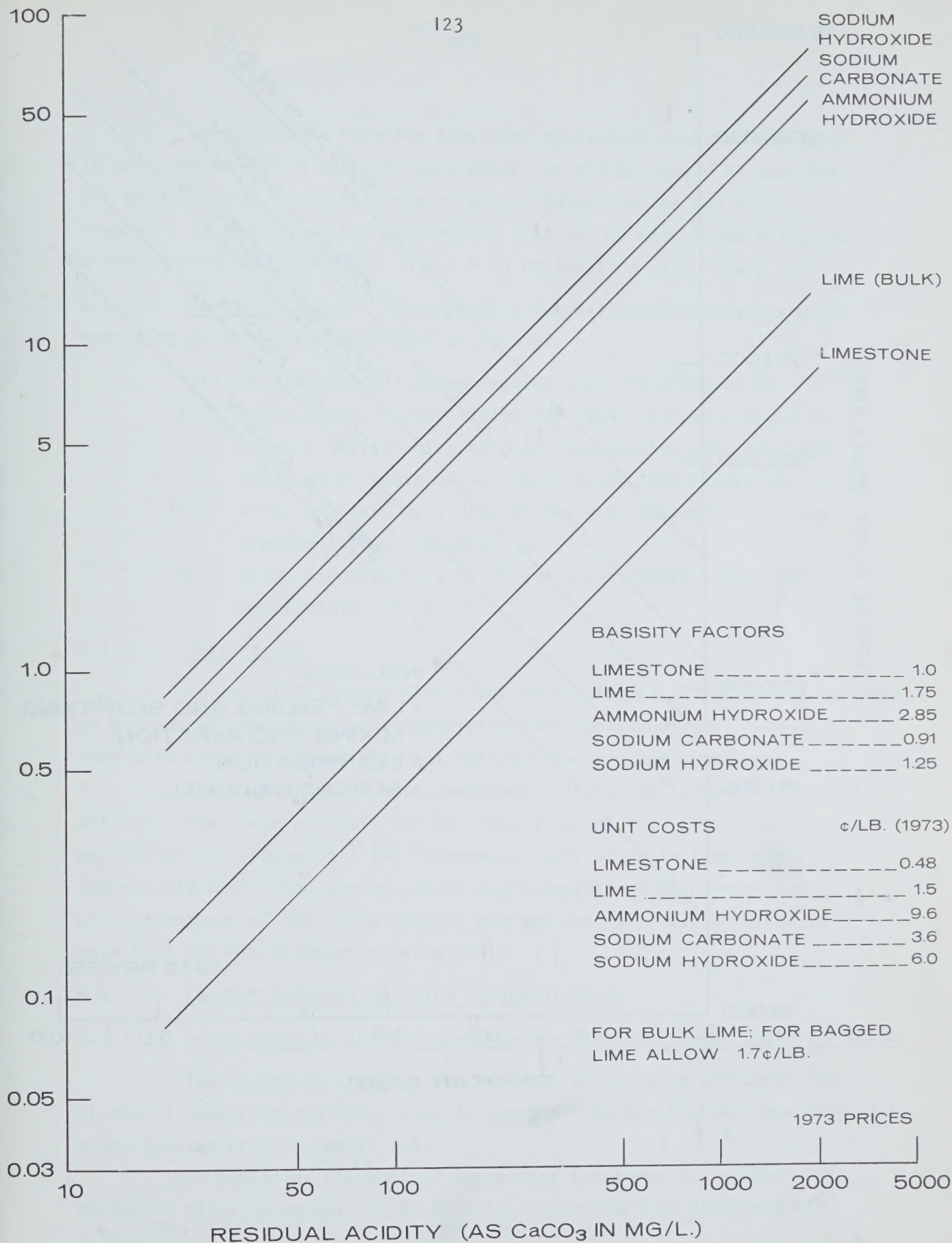


Figure 28. Neutralization Reagent Costs

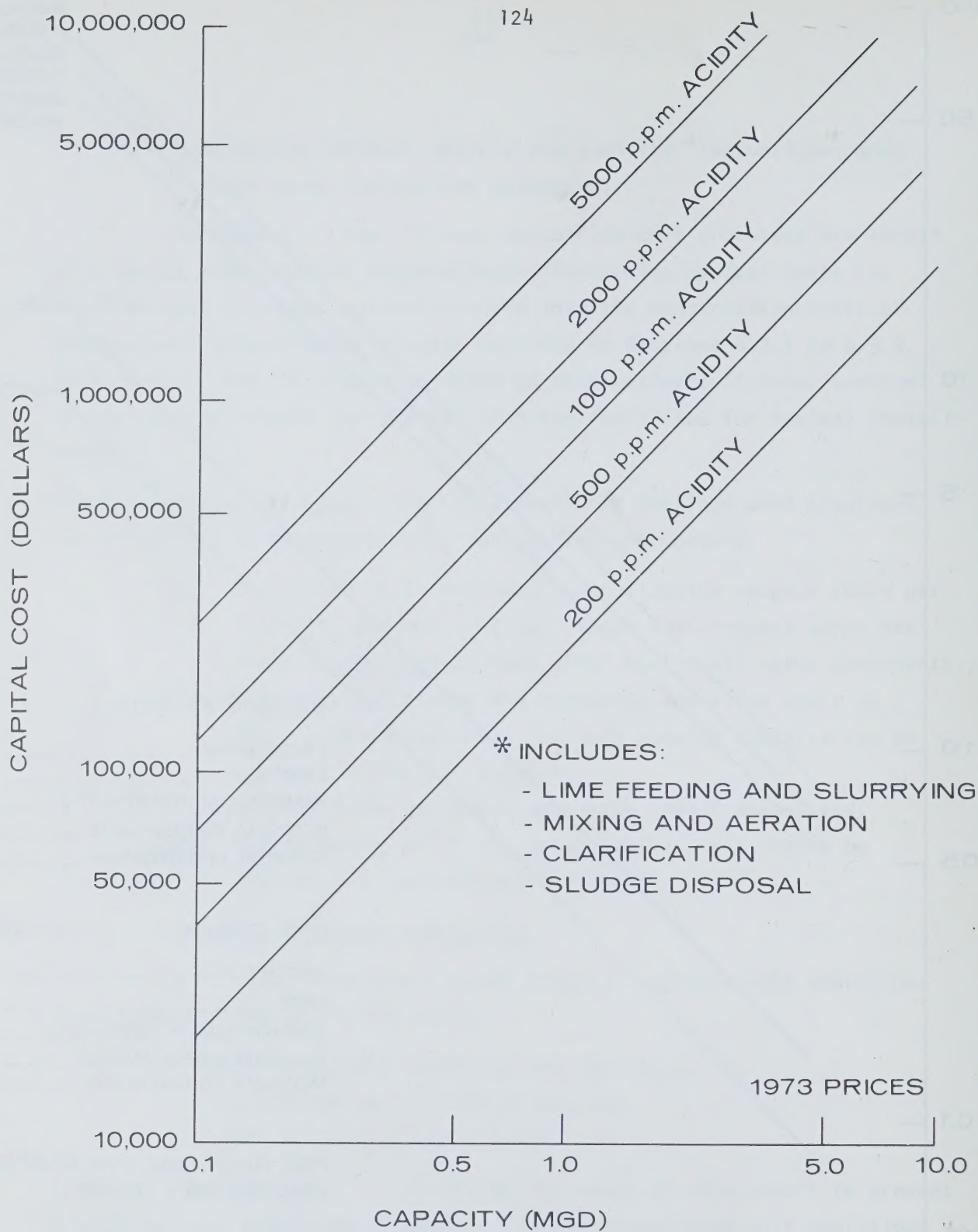


Figure 29. Capital Costs - Hydrated Lime Treatment Plant *

In some cases separate treatment operations might be employed in conjunction with a tailings pond which, in effect, would be used for the gross separation of tailings prior to neutralization and metals removal. In such cases the approximate cost can be calculated using the procedures in Sections 6.3.1 to 6.3.4 in conjunction with Figure 29.

6.4.2.2 Operating costs. The procedure for estimating treatment plant operating costs may be summarized as follows:

- (i) obtain neutralization reagent costs from Figure 28;
- (ii) allow approximately 125 hp per 1000 U.S. gpm treated as a basis for calculating power consumption and calculate total power costs on an hourly basis from Figure 23;
- (iii) allow approximately \$10,000 per year for each full time treatment plant operator; and,
- (iv) allow approximately 6% of the plant capital costs for maintenance and repair.

6.4.3 Total costs

To assist in determining total costs, Tables 14 and 15 are included as a means of condensing and summarizing each individual cost component. They also serve to insure that all relevant factors are considered.

The total cost of waste management in Canada is moderately well defined. The range of costs for tailings disposal is 0.5¢ to 35¢ per ton of ore processed. The incremental cost of waste treatment to upgrade old facilities appears to be approximately 10-20¢/ton of ore with an extreme of 35¢. The current average cost is 7.5¢/ton based on data from the Mining Questionnaire (16).

6.5 Further Examples of Waste Treatment Costs

6.5.1 Costs based on the Northeastern New Brunswick Water Quality Program

The following analysis for the cost of treating effluents from mine/mill operations is taken from Volume 3 of the Northeastern New Brunswick Water Quality Program report (18).

In assessing the cost of wastewater treatment it is important to define eligible costs. To do this it is necessary to differentiate between:

TABLE 14. CAPITAL COST ESTIMATION SUMMARY

DRAINAGE CONTROL

Contaminated drainage control ditches	\$ _____	
Holding pond dams	\$ _____	
Holding pond spillway structures	\$ _____	
Holding pond seepage collection ditches	\$ _____	\$ _____
Stream diversion channels	\$ _____	
Stream diversion dams	\$ _____	
Minor access bridges	\$ _____	
Culverts	\$ _____	\$ _____

PUMPING AND PIPING

Pipe material	\$ _____	
Pipe insulation	\$ _____	
Electrical tracing	\$ _____	
Clearing and grubbing	\$ _____	
Grade preparation	\$ _____	
Pipe installation	\$ _____	\$ _____
Pumps	\$ _____	
Electric motors	\$ _____	
Pump and motor installation	\$ _____	
Pumphouses	\$ _____	
Transmission lines	\$ _____	\$ _____

TAILINGS IMPOUNDMENT

Starter dams	\$ _____	
Dam construction	\$ _____	
Decant structures	\$ _____	
Spillways	\$ _____	
Seepage collection ditches	\$ _____	\$ _____

TAILINGS POND TREATMENT

Neutralization reagent storage and handling \$ _____

SEPARATE TREATMENT OPERATION

Hydrated lime treatment plant \$ _____

\$ _____

Sub-Total

\$ _____

Allow 30% contingency

\$ _____

Allow 10% engineering

\$ _____

Total order of
magnitude cost

\$ _____

TABLE 15. OPERATING COST ESTIMATION SUMMARY

PUMPING AND PIPING

Pump power costs \$ _____

Maintenance and repair \$_____

Pumphouse supervision \$ _____

\$ _____

TAILINGS POND TREATMENT

Neutralization reagent cost \$ _____

Operating labour \$ _____

Maintenance and repair \$_____

SEPARATE TREATMENT OPERATIONS

Neutralization reagent costs \$_____

Operating labour \$

Maintenance and repair \$

\$ _____

Total annual operating costs \$ _____

(a) those items which are basic practices in any well managed mine and have, as a side benefit, environmental protection implications - considered ineligible; and,

(b) those additional facilities and procedures which are primarily determined by environmental protection requirements - considered eligible.

As an example, in the analysis presented here the capital and operating cost of facilities to handle tailings disposal in a safe, engineered manner are considered normal mine practice and are not eligible as pollution abatement expenditures. However, where the size of the stilling basin has been increased or the use of an impervious embankment is indicated or provisions are made for metal hydroxide sludge storage in the tailings pond, then the incremental cost has been considered eligible.

In order to illustrate the costs associated with providing wastewater treatment facilities a case history approach has been used. In the following analysis costs have been calculated for both complex and simple effluent types for assumed mill capacities of 1000 ton/day, 500 ton/day and 10,000 ton/day.

It is assumed that the effluent requirements will be achieved by the provision and proper operation of well controlled lime neutralization and precipitation, together with adequate stabilization and sedimentation facilities. These may take the form of a relatively simple but well controlled tailings pond type of system or, particularly for more complex effluents, may consist of segregated treatment operations such as oxidation, neutralization, coagulation and sedimentation. The more complex and, therefore, costly extreme has been analyzed in this section with a view to determining the maximum incremental costs. But it must be realized that simpler systems may often suffice, depending on the following factors:

- type of milling process and its implications on recycle and waste treatability, particularly with respect to metal complexing, solubility relationships, and precipitate formation;
- average value and range of effluent volume as influenced by the above and also by the volume of mine water and contaminated surface drainage;

TABLE 16. DESIGN ASSUMPTIONS FOR COST EVALUATION

Parameter		1000 Tpd	5000 Tpd	10,000 Tpd
Process (500 gal/ton)	gpm	347	1736	3470
Assumed Mine Water	gpm	150	225	300
Process losses (10 gal/ton)	gpm	7	35	69
Loss to Tailings Moisture Content (85% Solids)	gpm	20	100	200
Volume for Treatment - LOW	gpm	470	1826	3501
Precipitation on Contaminated Areas (40"/year)	gpm	86	343	686
Evaporation from Pond Surfaces (18"/year)	gpm	23	139	294
Volume for Treatment - AVERAGE	gpm	532	2030	3893
5 1/24 hour storm balanced over 30 days	gpm	130	522	1044
Volume for Treatment - HIGH	gpm	662	22552	4937

Note: All gallons Imperial

- the degree of oxidation required depending on the chemistry of the waste, thiosalt concentrations, ferrous/ferric ratio, oxidizable reagent residuals and other compounds, etc.; and,
- the actual performance of various treatment configurations.

6.5.1.1 Effluent stabilization. Stabilization may be required for effluents containing thiosalts, ferrous iron and unconsumed oxidizable process reagents. Although process design requirements are difficult to specify it has been assumed that a six-day retention period would be required for a complex effluent and six hours for a simple one.

Costs for aeration have been based on provision of a two cell aerated lagoon of 15 ft depth assuming construction is on level ground and that dyke materials are readily available. The cost of aeration equipment is based on the air-gun type of installation. This form of aeration appears well suited to mine waste stabilization as it is not affected by acid conditions, has good winter operating characteristics, low operating and capital costs and is reported to be simple and reliable in operation. The assumed aeration operating parameters for the six cases under consideration are listed in Table 17.

6.5.1.2 Clarification including reagent addition. Actual cost data for reagent mixing and for clarifier/flocculator equipment are not readily available, although several generalized cost evaluation curves for these and other treatment operations exist and have been used to establish order of magnitude costs for the assumed design conditions under consideration. The main sources of cost data used for this section are Smith, R. (19); Evans, D.R. and Wilson, J.C. (20); and, Eckenfelder, W.W. Jr. and Ford, D. L. (21). All costs have been updated to 1972 dollars using the ENR costs index.

Although reagent costs may differ from the two mill types under consideration it is not possible to quantify the differences using available data; identical costs have, therefore, been assumed for both effluent types.

The capital and operating costs of clarification, flocculation, reagent supply, storage and mixing are commonly quoted without breakdown and are used in this manner here. The assumed design data and the resulting costs are indicated in Tables 17 and 18.

TABLE 17. AERATION PARAMETERS AND COSTS FOR AVERAGE FLOWS
(costs based on 1972 data)

Parameter	Unit	Simple Effluent			Complex Effluent		
		1000 Tpd	5000 Tpd	10,000 Tpd	1000 Tpd	5000 Tpd	10,000 Tpd
Retention	hours	6	6	6	144	144	144
Pond Value	cu ft	30,700	117,000	225,000	737,000	2,800,000	5,400,000
Surface Area 15' Depth	sq ft	2,050	7,800	15,000	49,000	187,000	360,000
Volume Diking	cu yd	4,200	8,200	11,400	20,600	40,200	56,000
Cost of Earthworks	\$	21,000	41,000	57,000	82,000	120,000	170,000
Assumed TOD	mg/l	100	100	100	570	570	570
Oxygen Requirement	lb/ton	21	105	208	120	600	1,185
Approx. No. of guns		9	42	84	48	240	474
Capital Cost of Aeration Equipment	\$	4,000	13,000	25,000	14,000	67,000	130,000
Air Requirement	Scfm	150	710	1,430	815	4,080	8,060
HP @ 50% Efficiency		15	50	100	60	300	600
Cost of Blower Installation	\$	19,200	25,000	32,000	25,800	59,000	97,000
Capital Cost of Equipment	\$	23,200	38,000	57,000	39,800	126,000	227,000
Operating Cost	\$/year	1,700	5,700	11,500	6,900	34,500	69,000
Total Construction Cost	\$	44,200	79,000	114,000	122,000	246,000	397,000
Amortized @ 10% over 15 years	\$/Year	5,800	10,400	15,000	16,000	32,300	52,200
Total Cost/Year	\$	7,500	16,100	26,600	22,900	66,800	121,200
Total Cost/Ton	¢	2.05	0.88	0.73	6.27	3.66	3.32

TABLE 18. CLARIFICATION AND REAGENT ADDITON COSTS
(costs based on 1972 data)

Parameter	Unit	Mill Capacity		
		1000 Tpd	5000 Tpd	10,000 Tpd
Design Flow - Average	gpm	532	2,030	3,893
- Peak	gpm	662	2,552	4,937
- Low	gpm	470	1,826	3,501
Clarifier Overflow Rate - Average	gpd/ft ²	750	750	750
- Peak	gpd/ft ²	933	942	951
- Low	gpd/ft ²	662	675	675
Clarifier Area Assumed Units	sq ft	1,021	3,900	7,470
		2@ 25 ft dia.	2@ 50 ft dia	2@ 70 ft dia
Estimated Capital Cost	\$	60,000	200,000	380,000
Estimated Operating Cost	\$/Year	12,800	47,300	87,700
Capital amortized over 15 years at 10%	\$/Year	7,880	26,300	50,000
Total Cost Per Year	\$	20,600	73,600	138,000
Total Cost Per Ton	¢	5.6	4.0	3.8

6.5.1.3 Total treatment costs. The total cost figures for the design conditions assumed are indicated in Table 19. With capital costs amortized at 10 per cent over 15 years, the maximum cost of providing the specified treatment downstream of the tailings pond is 14.8 cents/ton of ore processed. This does not constitute an incremental cost of providing wastewater treatment as many mining operations have already provided many of the facilities applicable to their particular operation.

These costs do not include provision for sludge handling and storage. The effect of this component will vary greatly and is proportional to the amount of dissolved heavy metals present in the wastewater. In an extreme case, the additional volume needed is 50% of that occupied by the tailings themselves. Estimates of the cost of sludge handling and storage range up to 20¢/ton of ore. This results in a net cost up to 35¢/ton of ore. The effect of storing sludge separately from tailings is not clearly defined but it is likely to be greater than for combined disposal. The reason for this is believed to be the increased flocculation and, thus, increased density of sludges formed in the presence of other solid materials such as tailings and limestone particles.

6.5.1.4 Advanced treatment costs. The possibility exists that, in certain very critical areas, it may be necessary to consider the treatment of mining effluents using one or another form of advanced treatment. Although the technology of such applications is not yet fully developed, and costs will undoubtedly fall once their full scale application has been established, it is quite likely that present operating costs could be in the order of \$1.00 per 1000 gallons or more, depending on the system used. The operating costs for ion exchange are reported as varying from 33 cents to \$1.00 per 1000 gallons in Table 8. The U.S. Dept. of the Interior reports a range of \$0.30 to \$2.53 for ion exchange in acid mine drainage applications (22). Thus, even assuming an optimistic value of 70 cents per 1000 gallons, the operating costs for the assumed 0.8 mgd_e effluent volume for the 1000 ton/day mill would be 56 cents per ton. Capital costs would probably be in the order of \$1 million if regeneration waste treatment were included and this would lead to amortization costs of 36 cents/ton or a total of 92 cents/ton.

TABLE 19. TOTAL TREATMENT COST ¢/TON PROCESSED
(1972 data)

Item	Simple Effluent			Complex Effluent		
	1000 Tpd	5000 Tpd	10,000 Tpd	1000 Tpd	5000 Tpd	10,000 Tpd
Oxidation	2.05	0.88	0.73	6.27	3.66	3.32
Clarification Including Reagent Additions, etc.	5.6	4.0	3.0	5.6	4.0	3.0
Sub-Total	7.65	4.88	3.73	11.87	7.66	6.32
Allow 25% for misc. piping, pumps, etc.	1.91	1.22	0.93	2.97	1.91	1.58
Total Cost ¢/Ton processed	9.56	6.10	4.66	14.84	9.57	7.90
¢/1000 gal treated	12.5	10.4	8.3	19.4	16.4	14.1

It is apparent that the costs of such advanced treatment are of a very different order to those associated with providing even the most sophisticated conventional methods, if the whole waste stream has to be treated. However, if only the average net effluent required advanced treatment the incremental cost over and above conventional treatment might be more in the order of 20 to 30 cents per ton, once the technical feasibility of such treatment has been established.

6.5.1.5 Overall impact of waste management. If the assumptions made in this somewhat brief analysis are in order, it may be concluded that the economic impact of incorporating the best available waste management practices at new mining developments are relatively small. Indications are that, on the average, waste minimization through surface drainage control and the provision of a maximum degree of process recycle will result in minimal additional expenditures and possibly even cost savings for new developments. There will undoubtedly be exceptions to this conclusion and the cost of incorporating waste minimization techniques at existing mines subsequent to development may be very costly.

The cost of providing a well controlled treatment system (consisting of aeration, neutralization and sedimentation), in addition to a tailings disposal area, for a small mill with complex effluent characteristics, may approach 35 cents per ton processed but will often be less than this, as the figure has been based on relatively conservative assumptions and include the provision of clarifiers when a well designed cellular lagoon system may be quite adequate. For mills with simpler effluent characteristics, costs should not exceed 10 cents per ton processed and again this represents the total costs of providing relatively sophisticated treatment downstream of the tailings system.

The foregoing analyses do not take into account the effects of taxation. However, it must be stressed that present taxation legislation permits an accelerated capital cost write-off for pollution control structures and equipment over two years.

The actual impact of the cost of waste management, if assumed to range between 10 to 35 cents per ton, is difficult to express in quantitative terms in relation to the profitability of a mining opera-

tion as operating costs vary widely and data on the profitability of specific operations are not readily available.

The profitability of a specific operation naturally depends on the grade of the ore (i.e., the amount of saleable metal realized per ton processed) and the contract price that the mine is able to negotiate for that metal in the form of concentrates to a smelter. This varies according to world metal prices.

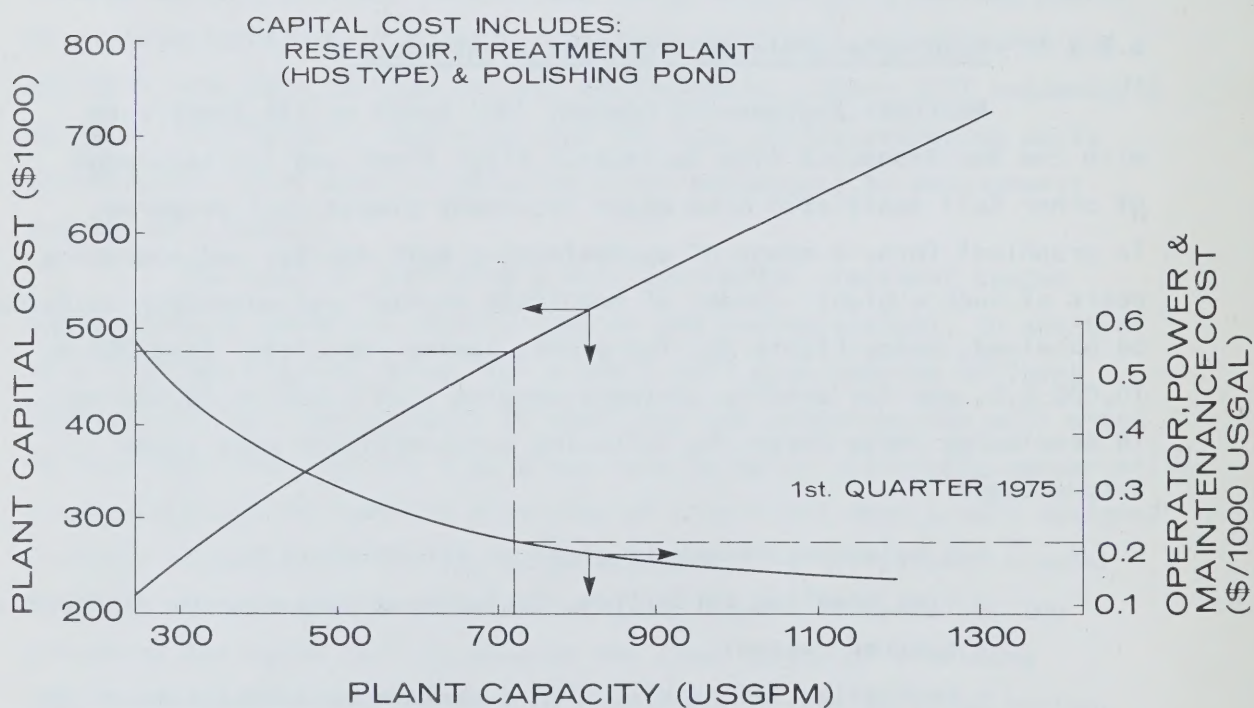
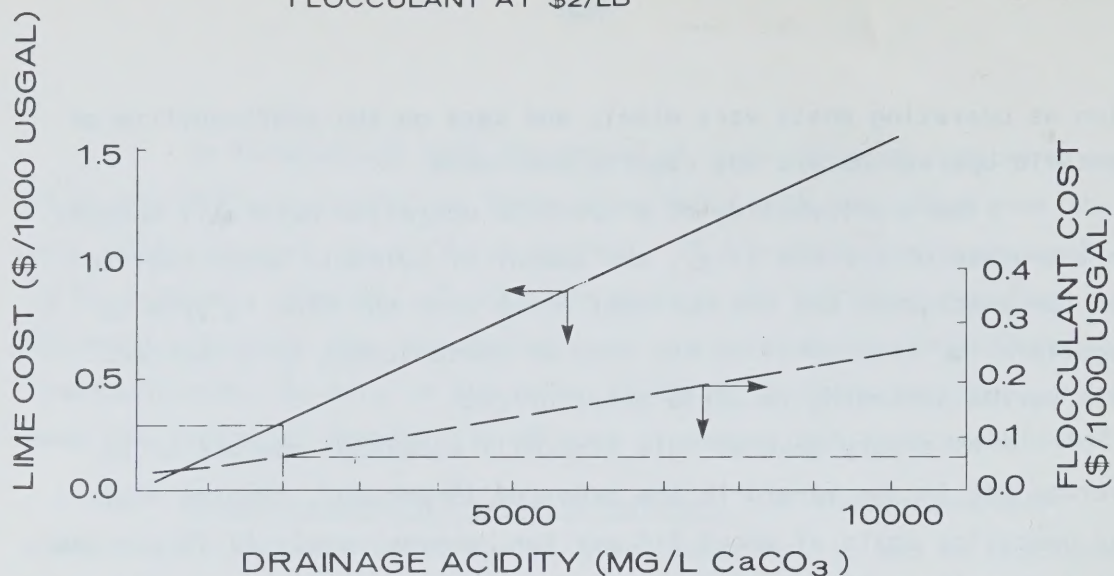
As a very general rule mine/mill complexes operate on an average net income margin in the order of 15 percent, meaning that for operating costs of about \$10 per ton, approximately \$1.75 per ton net income is realized and it is to this figure that the cost of waste management must be related. It can be seen that the costs detailed in this section range from three to twenty percent of the assumed net income.

6.5.2 Acid mine drainage treatment plant costs

Montreal Engineering Company (9), based on its experience with the New Brunswick Mine Wastewater Pilot Plant and its knowledge of other full scale acid mine water treatment plants, has prepared, in graphical form, a means of approximating both capital and operating costs of such a plant. Order of magnitude capital and operating costs can be obtained, using Figure 30, for plants having capacities from 250 to 10,000 U.S. gpm for acidity contents ranging from 1,000 to 10,000 mg/l. In developing these costs the following considerations were taken into account:

- a balancing reservoir of 24 hr retention time;
- lime handling facilities, including a lime storage silo and a pH control system;
- neutralization tank(s) with a maximum residence time of 30 minutes;
- flocculant preparation and feed system (a separate flocculation vessel was not included);
- lime/sludge flash mix reactor when recycle is used;
- a conventional clarifier - thickener;
- all necessary piping, pumps and mixers;

LIME REAGENT AT \$50/TON
FLOCCULANT AT \$2/LB



EXAMPLE:

FLOWRATE = 720 USGPM
ACIDITY = 2000 MG/L CaCO_3

CAPITAL COST = \$480000

OPERATING COST:

POWER, OPER., MAIN. = $\$.21/1000 \text{ USGAL}$

LIME = $\$.29/1000 \text{ USGAL}$

FLOCCULANT = $\$.07/1000 \text{ USGAL}$

TOTAL OPERATING COST = $\sim \$.57/1000 \text{ USGAL}$

Figure 30 Acid Mine Drainage Treatment Order of Magnitude Cost Estimate
(Does Not Include Sludge Dewatering and Disposal Cost)

- basic controls;
- building; and,
- discharge lagoon of 24 hr retention time.

Operating costs include one plant operator, power at \$0.02/kw hr, lime costs of \$50/ton and flocculant at \$2/lb.

Sludge dewatering and disposal costs are not included since these are very much dependent upon the disposal alternative available at a particular mine site.

As already indicated in Section 5.6 the capital and operating costs for the 200 lpm acid mine water treatment plant at Chester Mines in New Brunswick are \$110,000 (1974) and \$0.70/1000 Imp. gal of mine drainage treated.

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